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Old Dogs, New Tricks

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EDITORIAL

Gulf and Caribbean Research is pleased to announce migration to a fully on-line delivery platform with the publication of Volume 26. The peer-reviewed, scientific journal was originally founded in 1961 by Gordon Gunter as a publication of the Gulf Coast Research Laboratory and was titled *Gulf Research Reports*; that name persisted through 1999. Starting in 2000, the name was changed to *Gulf and Caribbean Research* to better reflect the scope of manuscripts, and the journal was published in traditional hard-copy format through 2013. Our new fully on-line delivery format is published by the University of Southern Mississippi on the *Aquila* platform, and started production in April 2015. In celebration of our new on-line format, manuscripts submitted by 31 August 2015 will be published in Volume 26 which will be completely open access at no charge to the author. Manuscripts submitted after this date will appear in subsequent volumes, which will be a hybrid on-line format. In the hybrid volumes, authors can chose to have their complete manuscript published as open access for a fee, or can elect to have only the abstract available on-line for no publication fee. A PDF of these latter papers will be available for a fee.

Gulf and Caribbean Research, now in its 54th year of publication, continues to consider manuscripts which deal mainly with research or research issues pertinent to the Gulf of Mexico and Caribbean Sea (<http://aquila.usm.edu/gcr/aimsandscope.html>). These include full papers, short communications, review papers, and book reviews. *Gulf and Caribbean Research* continues to provide an important outlet for peer-reviewed publications from the Gulf of Mexico and Caribbean Sea regions. The same reviewing and publishing standards that we have used in the past will be used for our new on-line format. We anticipate that more papers will be published per year with a shorter turn-around time with this new format.

Details of the journal history can be found in the links below:

<http://aquila.usm.edu/gcr/about.html>

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Movement of Bigmouth Sleeper, *Gobiomorus dormitor*, in the Río Cañas, Puerto Rico, revealed by radio telemetry, and a discussion of the species' amphidromous characterization

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MOVEMENT OF BIGMOUTH SLEEPER, *Gobiomorus dormitor*, IN THE RÍO CAÑAS, PUERTO RICO, REVEALED BY RADIO TELEMETRY, AND A DISCUSSION OF THE SPECIES' AMPHIDROMOUS CHARACTERIZATION

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ABSTRACT: Bigmouth Sleeper, *Gobiomorus dormitor*, are tropical fish native to rivers of Puerto Rico. They are popular sport fish targeted by local anglers. They are considered as diadromous, but this characterization is controversial. The displacement of Bigmouth Sleeper in the Río Cañas, Puerto Rico, was examined using radio telemetry. Twenty fish were radio-tagged and monitored from January to November 2008. Fish were in the Río Cañas 69%, 62%, and 59% of the time before (January–May), during (June–August), and after (September–November) the spawning season, respectively. Average detected linear home range (within the river) for all tracking periods was 2.3 km, but varied from less than 0.1 km to 8.1 km. Detected linear home range was not related to weight class or sex. They may remain in freshwater as juveniles and adults, only spending their larval stage in estuarine or marine habitats (i.e., amphidromous diadromy). However, they can complete their larval stage in freshwater but appear to make periodic migrations to the marine environment as adults, as suggested by this study. The best descriptor of Bigmouth Sleeper is that of a facultative amphidromous euryhaline species. Fishery and habitat management for this fish should take into consideration their general migratory behavior and their plasticity with respect to this life history strategy.

KEY WORDS: amphidromy; eleotrid; euryhaline; migration

INTRODUCTION

Bigmouth Sleeper, *Gobiomorus dormitor*, is a tropical fish native to the Caribbean Basin and Central America (Gilmore 1992), where they play an important role in sport fishing. They are euryhaline and inhabit coastal lagoons and freshwater rivers (Gilmore 1992, Hernández–Saavedra et al. 2004). In Puerto Rico, Bigmouth Sleeper is the only native riverine fish species that grows well in artificial impoundments (Neal et al. 2009). The Puerto Rico Department of Natural and Environmental Resources (DNER) expressed interest in using Bigmouth Sleeper as a native alternative to the current introduced reservoir sport fish assemblage.

Information on life history characteristics and migratory behavior of Bigmouth Sleeper is limited and the data are conflicting. They have been classified as anadromous (Nordlie 1979), catadromous (Nordlie 1981, Gilmore and Hastings 1983), and amphidromous (Winemiller and Ponwith 1998). Hernández–Saavedra et al. (2004) were unable to determine spawning location for Bigmouth Sleeper in the Tecolutla estuarine system of Mexico. Smith and Kwak (2014) reported that most Bigmouth Sleeper in Puerto Rico rivers were amphidromous. However, a small proportion of Puerto Rico Bigmouth Sleeper deviated from classical amphidromy (Smith and Kwak 2014). Bacheler et al. (2004) reported a self-sustaining population in Carite Reservoir, Puerto Rico, and several researchers have indicated landlocked populations in natural lakes of Honduras (Darnell 1962) and Nicaragua (McKaye et al. 1979, Bedarf et al. 2001).

Management of a sport fishery requires detailed knowl-

edge of the species' life history. This is especially important for riverine species demonstrating diadromous behaviors in systems fragmented by artificial barriers (Cooney and Kwak 2013). Since the Bigmouth Sleeper has some degree of plasticity in their life history strategy, we hypothesize if freshwater habitats are limiting or become limited, the Bigmouth Sleeper might respond by spending a greater proportion of time in an adjacent marine habitat. The goal of this study was to describe movement patterns of Bigmouth Sleeper in a Puerto Rico river and to reconsider their diadromous characterization.

MATERIALS AND METHODS

Study area

Puerto Rico is the smallest and easternmost island of the Greater Antilles. The Río Cañas is a non-navigable river that flows north to south through the coastal city of Ponce (Figure 1). This river was chosen for study because it has a known population of Bigmouth Sleeper and has a large waterfall at river kilometer (Rkm) 13. The waterfall effectively limits upstream movement and provides a manageable river reach for telemetry research. The river is shallow (~ 0.6 m deep on average, with some pools to 1.5 m) and narrow (~20 m across at the mouth), with a mean \pm sd discharge of $14.4 \times 10^6 \pm 6.09 \times 10^6$ m³/yr.

Transmitter implantation

Lotek model NTC–6–1 transmitters (22.4 x 9.1 mm; 2.8 g; 18 cm antenna) were used for this study. Four trans-

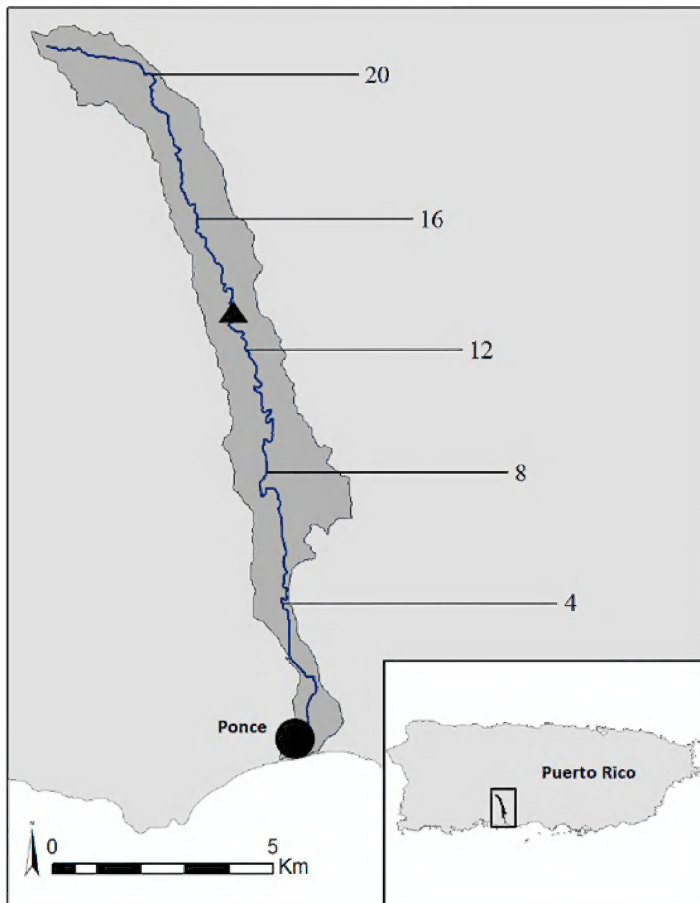


Figure 1. Map of the Río Cañas basin and watershed, and insert showing location on the south coast of Puerto Rico. Triangle is the location of a waterfall, which inhibits upstream fish passage. Black circle is the city of Ponce. River kilometer is indicated.

mitters broadcast on a single frequency, with five different frequencies represented among the 20 transmitters used in this study (Lotek Wireless, Newmarket, Ontario, Canada). Transmitter life was projected at 357 d with a 10 sec signal burst interval. This transmitter model had a mass less than 2% of fish body mass (Winter 1996, Bacheler 2002).

A tag retention study was conducted in July 2007 to validate the implantation protocol. Ten Bigmouth Sleeper (females = 3; males = 7) from the Río Cañas were tagged during the retention study. Fish were anesthetized using an 80 mg/L dose of buffered tricaine methanesulphonate. Each fish was sexed according to Bacheler (2002), measured (TL, mm) and weighed (g). Anesthetized fish were placed in a grooved tray and water was continuously passed over the gills. Betadine® was applied to the surgery site. A small incision (~1 cm) was made anterior to the anus between the ventral side of the fish and the lateral line. The incision site was chosen to avoid contact with the substrate, as they are benthic fish. Sham Lotek model NTC-6-1 transmitters were used for the retention study. Each transmitter was immersed in a Betadine® bath prior to implantation. Transmitters were implanted in the abdominal cavity through the incision. A separate exit site for the external antenna was

created by inserting a 16 gauge needle through the abdominal wall between the incision and the anus. The external antenna was threaded through the bore of the needle and consequently through the abdominal wall. The 16 gauge needle was then removed from the abdominal wall and slid off the external antenna. The incision was closed with three 3/0 Monoswift monofilament absorbable sutures using a 26 mm tapered needle. A biological adhesive was also applied to the wound. Surgical instruments were sanitized between surgeries (Winter 1996, Bacheler 2002). The mean weight of tagged fish was 224 ± 73.5 g and mean surgery time was 5.6 ± 1.4 min. Fish were held for 30 d during the tag retention study. After 30 d, there was a 20% tag loss rate. There were no mortalities during the tag retention study.

Specimens for the actual telemetry study were collected from 3 sites on the Río Cañas between Rkm 5 and 8 using backpack electrofishing on 12–17 December 2007. Fish were held at DNER's Maricao Fish Hatchery in two 1,552 L polyethylene tanks at half volume prior to and following transmitter implantation. Systems utilized flow-through water derived from the Maricao River. Radio transmitters were implanted in Bigmouth Sleeper (females = 5, males = 15) on 20 December 2007 using the protocol outlined above. Experimental fish had a mean weight of 250 ± 108 g and a mean TL of 316 ± 46 mm. All tagged fish were at least 200 mm TL, which is above the reported minimum length at maturation (Bacheler 2002). Fish were held at the Maricao Fish Hatchery for 14 d post-surgery and monitored for incision healing and tag retention. For convenience, the 20 fish were numbered 11 through 30.

Radio telemetry

On 4 January 2008, 5–8 tagged fish were randomly chosen for release at each of the original collection sites. Transmitters were operational in all 20 fish at the time of release. A tracking period was the 1–2 d interval during which the length of the river was searched for radio signals. Tracking was conducted during daylight hours, on foot, by slowly wading downstream from the upstream most location while scanning the appropriate radio frequencies. A Garmin GPS unit was used to determine the location of individuals as the fish were detected. The first tracking period was 7 January 2008. Tracking periods occurred about every other week from 7 January to 30 May 2008. Tracking periods occurred weekly during the presumed spawning season (June to August; Harris et al. 2012). Tracking periods reverted to a frequency of every other week from 12 September to 21 November 2008.

Non-detection of a radio signal could indicate transmitter failure, angler harvest of a tagged fish, a false non-detection (i.e., failure to detect the fish despite its presence in the Río Cañas), or fish movement out of the study area. A non-detection during one tracking period followed by detection during a subsequent tracking period would rule out transmitter failure or angler harvest. At a walking pace of

about 1.6 km/hr and a transmitter range of about 0.7 km (based on the exponential model of Freund and Hartman (2002) and a maximum depth of 1.5 m in the Río Cañas), the tracker would be in transmitter range for ~52 min. The receiver scanned each of the 5 transmitter frequencies for 10 sec. The receiver would have had to miss the signal more than 70 times, while in range of the transmitter, to create a false non-detection during a tracking period. Few radio telemetry studies quantify rates of false non-detection, or conversely, rates of detection. Enders et al. (2007) reported detection rates of $99.3 \pm 2.2\%$ and $96.9 \pm 6.5\%$ for radio tagged juvenile Atlantic Salmon, *Salmo salar*, and Brown Trout, *Salmo trutta*, during two study periods in Stoney River, Newfoundland, Canada. The shallow nature of the Río Cañas, the absence of side channels in the study reach, the characteristics of the transmitters, the method of tracking, and reported radio telemetry detection rates all suggest that false non-detections did not strongly influence this study.

Radio transmitters have conductivity restrictions, but the signals should be clear in relatively shallow water (< 4 m) with conductivity < 800 $\mu\text{S}/\text{cm}$, but become less detectable after this conductivity threshold is reached (Peters et al. 2008). Measurements taken in February 2007 about 1.25 km upstream of the Río Cañas mouth indicated that conductivities were acceptable for radio signal transmission (583 $\mu\text{S}/\text{cm}$; C. Mace, Mississippi State University, unpublished data). Conductivity measurements at the Río Cañas mouth varied from 3,940 – 50,500 $\mu\text{S}/\text{cm}$ in June–August of 2008 (Adelsberger 2009). It is most likely that signal non-detection during a tracking period indicated movement of a fish into high conductivity water near or seaward of the river mouth, rather than some other explanation. Data from this study were analyzed accordingly.

Data processing and analysis

Data for individual fish were truncated to include only the last detection of each fish and data from all preceding tracking periods. We refer to the length of time from release to last detection as the tracking duration. Non-detection during a specific tracking period within a truncated data set was interpreted as the fish being absent from the freshwater portion of the Río Cañas during that tracking period. The percent of time fish were present in the Río Cañas before, during, and after the presumed spawning season was estimated. We determined the average location of individual fish detected during each month of the study and then calculated the average location (in Rkm) of the group of fish during each month of the study. We also calculated the change in average location of the group between months of the study. Linear home ranges were calculated for each fish two ways. The detected linear home range represents the distance between the two farthest apart points where fish were actually observed. The presumed linear home range included Rkm 1 (where conductivity was too high for radio

signal detection) for fish absent for at least one tracking period. Hence, the presumed linear home range represents the distance between the upstream most sites where fish were observed and Rkm 1. Fish were grouped by weight using 100 g intervals and sorted by sex. Detected and presumed linear home ranges were tested for normality with Shapiro–Wilkes test and homogeneity of variance with Levene’s test (PROC UNIVARIATE; SAS Version 9.2) and data were rank transformed prior to further statistical analyses if they failed to meet these assumptions. The influences of weight class and sex on detected and presumed linear home ranges were examined using two-way analyses of variance (PROC ANOVA; SAS Version 9.2). The interaction terms between the two main effects variables were also tested for significance. An alpha level of 0.05 was used to determine significance in these statistical tests.

RESULTS

Out of the 20 fish tagged in the study, 2 were lost within the first 4 months (Figure 2). Fish 24 was caught in mid-March by a freshwater angler who reported the harvest and returned the transmitter to DNER. No other fish were reported as harvested. Nine fish were detected as late as October and November of the study. The number of fish detected

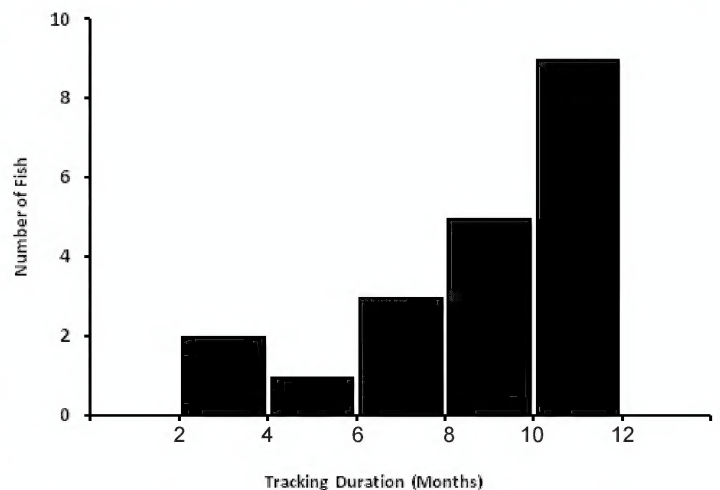


Figure 2. Tracking duration frequency distribution for Bigmouth Sleeper during a radio telemetry study in Río Cañas, Puerto Rico. Tracking duration is the time (in months) from release to the last detection of a fish during any tracking period.

during a tracking period generally declined throughout the study (Figure 3). The mean number of fish detections per tracking period was 8.7 ± 4.1 fish. Only 3–4 fish were detected per tracking period between 12 September and 31 October. However, an increase in fish detected per tracking period occurred in the last two tracking periods.

All 20 fish were detected during some portion of the pre-spawn period (January–May). These fish were in the Río Cañas an average of $69 \pm 32\%$ of the time (Figure 4). Of the 17 fish detected during some portion of the spawning season,

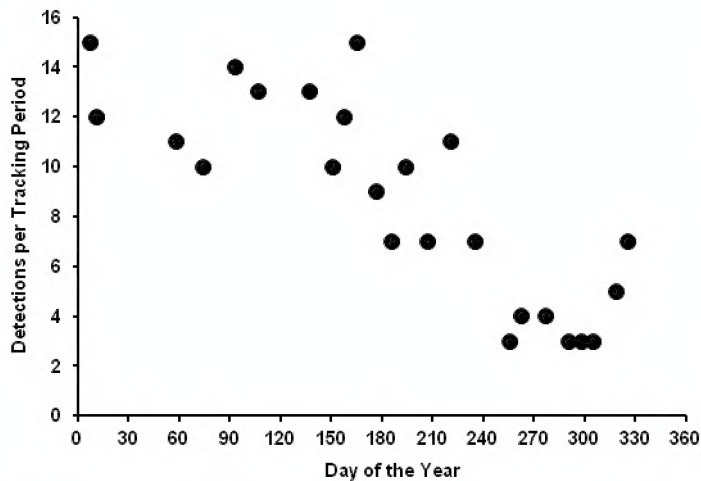


Figure 3. Number of Bigmouth Sleeper detections per tracking period during a radio telemetry study in Río Cañas, Puerto Rico. Each tracking period was a 2 d walking survey of the entire 13 km reach, during which detections and locations were recorded.

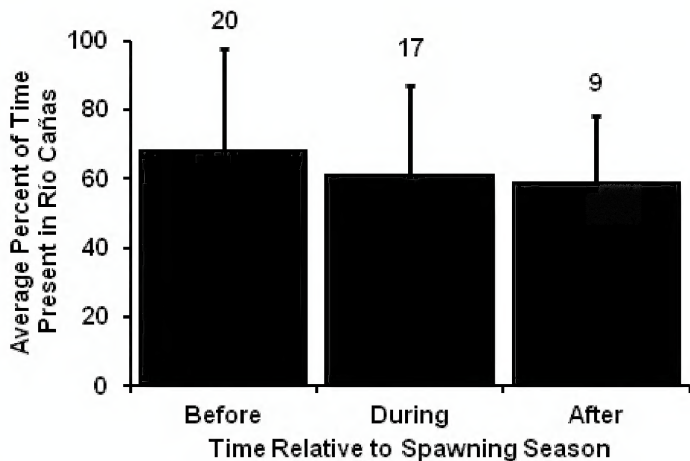


Figure 4. Average percent of time Bigmouth Sleeper were present in the Río Cañas, Puerto Rico, before (January–May), during (June–September), and after (October–November) spawning. Error bars represent one standard deviation. Number above the error bar represents the sample size.

fish spent an average of $62 \pm 26\%$ of the time in the Río Cañas. Nine fish were detected during the post-spawning period (September–November), and were in the Río Cañas an average of $59 \pm 19\%$ of the time.

Considered as a group, the fish showed little tendency to move up or down stream. The average location (in Rkm) of the group remained somewhat constant for the first 8 months of the study (Figure 5a). Between August and September, the average location of the group was displaced downstream about 3.6 km (Figure 5b). Between September and October, the average location of the group moved upstream about 1.5 km. No coordinated downstream movement by the group was detected during the presumed spawning period (Figure 5b).

Home ranges within the Río Cañas varied considerably among individuals throughout the study. Detected linear

home range averaged 2.3 ± 2.2 km. The smallest detected linear home range was less than 0.1 km (Fish 19, Figure 6), but this individual was only tracked for a little more than 2 months. The largest detected home range was 8.1 km (Fish 26), occupied during 9 months of tracking. Presumed linear home range averaged 5.7 ± 2.4 km. The largest presumed linear home range was 8.2 km (Fish 30, Figure 6).

The distributions of detected and presumed linear home ranges were not normally distributed (Shapiro–Wilk, $W = 0.084$, $p = 0.004$ and $W = 0.850$, $p = 0.005$, respectively). Variances for detected and presumed linear home ranges were homogeneous among weight classes (Levene’s test, $F_{2,17} = 1.70$, $p = 0.212$ and $F_{2,17} = 0.84$, $p = 0.448$, respectively). Variances for detected and presumed linear home ranges were also homogeneous between the sexes (Levene’s test, $F_{1,18} = 0.62$, $p = 0.442$ and $F_{1,18} = 1.54$, $p = 0.230$, respectively). The interaction term was not significant in either of the analyses of the effects of weight class and sex on detected and presumed linear home ranges. The interaction terms were removed from both models. The effect of weight class was insignificant in the analyses of detected and presumed linear home ranges, respectively (two–way ANOVA, $F_{2,16} = 0.86$, $p = 0.443$ and $F_{2,16} = 0.20$, $p = 0.822$). Likewise, the effect of sex was insignificant in the analyses of detected and presumed linear home ranges, respectively (two–way ANOVA, $F_{1,16} = 1.99$, $p = 0.178$ and $F_{1,16} = 0.02$, $p = 0.878$).

DISCUSSION

We failed to observe a coordinated movement by this group of tracked fish toward the mouth of the Río Cañas during the spawning season. Bigmouth Sleeper have been characterized as amphidromous (Winemiller and Ponwith 1998, Smith and Kwak 2014). Amphidromous species would presumably move downstream during the spawning season to facilitate rapid conveyance of eggs and larvae to the sea. We observed a downstream movement, but not until September, after the spawning season for Puerto Rico Bigmouth Sleeper identified by Harris et al. (2012). However, the spawning season might be longer than originally determined by Harris et al. (2012). Perhaps movements downstream were brief and episodic, and thus not observed during our 1–2 d tracking periods occurring weekly during the spawning season. Thus, our tracking data do not definitely prove the amphidromous characterization of this species.

The average detected linear home range was rather large for a stream fish. Home range studies of stream fish usually indicate little movement (30–122 m) with a trend showing larger home ranges for larger individuals (Gerking 1953). Gerking’s work led him to develop the “restricted movement paradigm” in stream fish ecology (Gerking 1959), though the paradigm has been challenged for some larger mobile fishes (Gowan et al. 1994). Home ranges are larger for species that migrate to spawn, but we note that home ranges for Big-

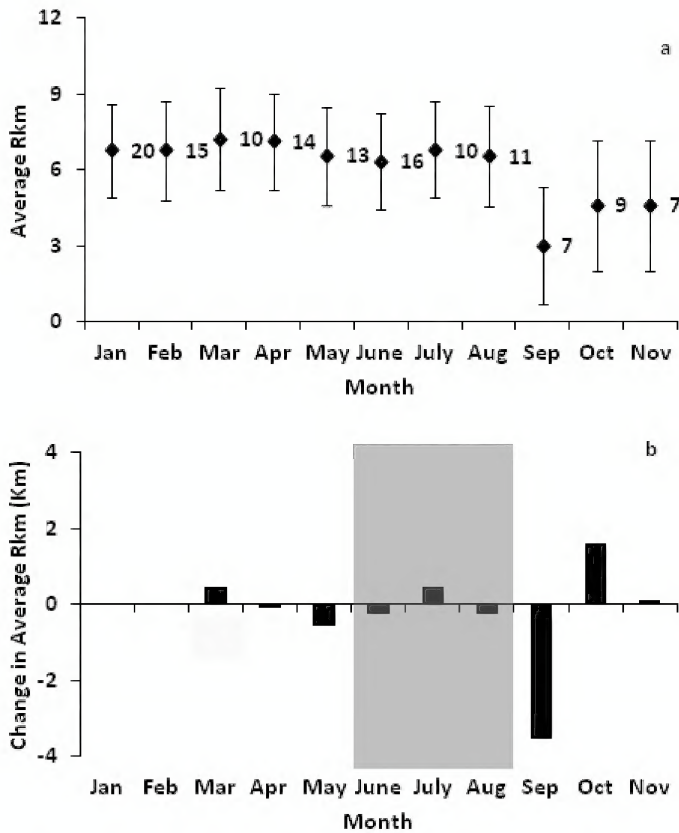


Figure 5. Location of Bigmouth Sleeper in Río Cañas, Puerto Rico. *a.* Average location (in river km, Rkm) of all Bigmouth Sleeper by month of the study. Error bars represent one standard deviation. Numbers next to the diamond indicate the number of fish detected during the month *b.* The change in average location (km) between months of the study. Negative numbers represent downstream movement of the average location. The shaded portion indicates the presumed spawning season.

mouth Sleeper in the Río Cañas appear particularly large, because of presumptive movements outside of the spawning season. Furthermore, if they spent time in coastal or marine waters adjacent to the Río Cañas, as these telemetry data suggest, then home ranges were likely larger than characterized here.

The restricted freshwater range in the Río Cañas could have influenced frequency and duration of the use of marine and estuarine habitats. Two of the 5 rivers studied by Smith and Kwak (2014), the Río Grande de Añasco and the Río Grande de Manatí, were large systems, with watershed areas of 468 and 471 km², respectively, and main channel lengths of 155 and 224 Rkm, respectively. Conversely, the Río Cañas watershed is less than 24 km², with a main channel length of 23 Rkm, and only 13 Rkm of accessible habitat due to the waterfall that acts as a barrier to upstream movement. A Bigmouth Sleeper should be able to traverse the entire length of the river in only a few hours.

In addition to stream length, stream discharge may affect the life history strategy of Bigmouth Sleeper. Because the Río Grande de Añasco and the Río Grande de Manatí are

larger systems, they have greater mean discharges. In addition, the Río Cañas drains the south coast of Puerto Rico, which is relatively dry compared to the rest of the island. For instance, the city of Ponce, which contains the lower portion of the Río Cañas, receives a mean annual rainfall of 97.7 cm/yr. By comparison, Mayagüez and Manatí, the cities at the mouths of the 2 largest rivers studied in Smith and Kwak (2014), receive mean annual rainfall amounts of 151.0 and 156.6 cm/yr, respectively (Current Results 2015). The Río Mameyes and the Río Sabana, the other 2 rivers studied by Smith and Kwak (2014), drain terrain that includes tropical rainforest. Although similar in watershed size and main channel length to the Río Cañas, these 2 systems likely experience more consistent base flow due to more consistent rainfall patterns. Thus, small size and irregular precipitation patterns may create variability in hydrology that is more pronounced in the Río Cañas than in these other Puerto Rico rivers. We note that daily discharge in the Río Cañas exceeded the average daily discharge plus 2 times the standard deviation 3 d in September and 7 d in October 2008. Such extreme flows could have displaced Bigmouth Sleeper from the river. When conditions in freshwater are not favorable (i.e., extreme low or high flow) and the distance to the marine environment is modest, it may be advantageous for fish to move to a more stable estuarine or marine habitat.

These data do not support a simple form of diadromy for Bigmouth Sleeper. Catadromous fishes spend their adult lives in freshwater and move to marine habitats to spawn. Anadromous fishes spend their adult lives at sea and move to freshwater to spawn (Myers 1949). Our data, though imperfect, suggest that Bigmouth Sleeper in the Río Cañas exhibited euryhaline salinity tolerance, spending the majority of their time in freshwater, but moving periodically between freshwater and estuarine or marine habitats.

Myers (1949) described amphidromy as a third form of diadromy where the migration from freshwater to marine

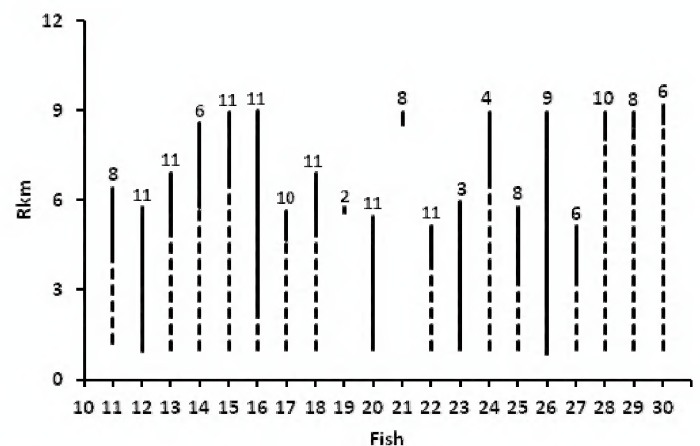


Figure 6. Detected (solid line) and presumed (solid line plus dashed line) linear home ranges of radio-tagged Bigmouth Sleeper. Number above home range is the number of months each fish was tracked.

habitat is “not for the purpose of breeding, but occurs regularly at some other definite stage of the life cycle.” McDowall (2007) refined Myer’s definition of amphidromy to include (a) spawning and hatching in freshwater, (b) rapid movement of larvae to sea, (c) feeding and growth at sea for a short period of time, (d) return to freshwater as a juvenile, and (e) feeding and growth in freshwater prior to maturity and spawning. Winemiller and Ponwith (1998) speculated that Bigmouth Sleeper and other Costa Rican eleotrids spent most of their time in freshwater but moved downstream to spawn in freshwater or upper estuarine habitats so that their larvae could occupy lower estuarine and marine habitats. This is consistent with McDowall’s definition of amphidromy. Larval development of Bigmouth Sleeper is characteristic of marine pelagic development: undeveloped mouth, small size, large yolk sac, and unpigmented eyes (Harris 2007). Furthermore, larvae were collected only at the river mouth of the Río Cañas during larval sampling throughout a 9 km reach in June–August 2008 (Adelsberger 2009, Neal et al. 2011). All of these observations are consistent with an amphidromous characterization. However, data from this study suggested periodic movements into estuarine or marine habitats. Furthermore, Bigmouth Sleeper are known to spawn in freshwater systems, such as Carite Reservoir, Puerto Rico (Bacheler et al. 2004), and Lake Jiloá, Nicaragua (McKaye et al. 1979). Marine development of larvae is not possible in these systems due to fish passage barriers. Out of necessity or choice, some Bigmouth Sleeper exhibit behavior that does not reflect amphidromy.

Data from this study appear to support the notion that Bigmouth Sleeper practice partial migration. Chapman et al. (2012) reviewed the concept of partial migration where some fish exhibit migratory behavior while others within the same population do not. Chapman et al. (2012) described partial migration as a life–history polymorphism, such that there is a degree of plasticity in terms of reproductive life history strategy among individuals within a species. The proportion of individuals adopting a specific life history strategy could depend on environmental conditions or constraints encountered during the spawning season. Smith and Kwak (2014) used otolith microchemistry to characterize Bigmouth Sleeper from Puerto Rico rivers as amphidromous. Microchemistry analyses of otoliths suggested a marine larval phase, followed by a solely freshwater adult phase for 87% of the fish they examined. However, Smith and Kwak (2014) used the phrase semi–amphidromous to characterize the 9.3% of individuals with otolith microchemistry indicating a freshwater larval phase and the 3.7% of individuals with otolith microchemistry indicating movements to marine habitats during their adult phase. Smith and Kwak (2014) speculated that otolith microchemistry indicating adult movement to a marine habitat could also be explained by a marine diet while in freshwater, rather than from actual residence in a

marine environment. Our Río Cañas data suggest that Bigmouth Sleeper periodically reside in marine habitats during their adult lives. The Río Cañas data do not contradict Smith and Kwak (2014), but they do indicate the proportion of fish moving between freshwater and marine or estuarine habitat during the adult phase is higher in the Río Cañas than the general estimate of 3.7% from Smith and Kwak (2014).

Exact characterization of the migratory behavior of Bigmouth Sleeper might be a matter of semantics. Smith and Kwak (2014) seem to be the first to use the term semi–amphidromous to describe them, citing Cronin and Mansueti (1971), Secor and Kerr (2009), and Whitfield (2005) for that terminology. Cronin and Mansueti (1971) refer to Striped Bass, *Morone saxatilis*, and White Perch, *M. americanus*, as semi–anadromous, stating, “Both of these are semi–anadromous fish, which move from saline water to, or almost to, freshwater for spawning.” Secor and Kerr (2009) define semi–anadromous as “diadromous fishes that spend most of their lives in saline water and that migrate to, or almost to, freshwater for spawning” and cite Cronin and Mansueti (1971) for that terminology. Secor and Kerr (2009) define semi–catadromous as “diadromous fishes that spend most of their lives in freshwater and that migrate to the estuary to breed” and cite Whitfield (2005) for that terminology. Semi can mean “half” or it can mean “partially or incompletely.” However, it is not clear from the phrase “semi–amphidromous” what part of the McDowall (2007) definition of amphidromy is or is not exhibited by semi–amphidromous fishes. Does the species spawn in freshwater, but not spend the larval phase in a marine habitat? Does a semi–amphidromous species spend the larval phase in a marine habitat, but does not spend its adult life solely in freshwater? Semi–amphidromous could mean that half of the individuals are amphidromous and half are something else. The term is ambiguous.

Metcalf et al. (2002) and McDowall (2007) used the term facultative, and Chapman et al. (2012) explicitly used the term “facultative amphidromy.” The phrase facultative means “capable of but not restricted to a particular function or mode of life.” Smith and Kwak (2014) reported that 87% of the Bigmouth Sleeper examined had otolith microchemistry indicating a marine larval phase followed by a solely freshwater adult phase. However, Smith and Kwak (2014) presented data suggesting that 3.7% of fish examined experienced marine or estuarine conditions during their adult phase. The telemetry data from the present study, though imperfect, suggest adult Bigmouth Sleeper do not remain solely in freshwater before, during, or after the spawning season. Adelsberger (2009) collected Bigmouth Sleeper larvae only at the moderately saline mouth of the Río Cañas. However, Smith and Kwak (2014) report that otolith microchemistry indicated that 9.3% of Bigmouth Sleeper examined had

freshwater larval phase. Carite Reservoir in Puerto Rico has juveniles that could not have spent their larval phase in marine water (Bacheler et al. 2004). These data lead us to favor the term facultative amphidromous euryhaline species to describe Bigmouth Sleeper.

This study suggests that Bigmouth Sleeper in the Río Cañas spent the majority of their time in the freshwater portion of the river. Nevertheless, they appeared to move between freshwater and estuarine or marine habitats throughout the year, and this movement has important consequences. First, fishery statistics, such as exploitation, should consider both marine and freshwater harvest by recreational anglers. Second, conservation efforts must

consider movement patterns and connectivity between the marine and freshwater habitats used by Bigmouth Sleeper. Natural impediments to movement, such as siltation at the river mouth, and anthropogenic impediments to movement, such as those described by Cooney and Kwak (2013), should be reduced or eliminated to allow migratory behavior of this and other Puerto Rico species. Finally, a careful examination of the landlocked population in Carite Reservoir might indicate whether that Bigmouth Sleeper population is merely exhibiting plasticity in its reproductive life history strategy, or if selective pressures are resulting in genetic changes towards a truly freshwater life history strategy.

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LITERATURE CITED

- Adelsberger, C.M. 2009. Natural life history characteristics of bigmouth sleeper *Gobiomorus dormitor* in Puerto Rico rivers. M.Sc. thesis. University of Arkansas at Pine Bluff, Pine Bluff, AR, USA, 70 p.
- Bacheler, N.M. 2002. Ecology of bigmouth sleeper *Gobiomorus dormitor* (Eleotridae) in a Puerto Rico Reservoir. M. Sc. thesis. North Carolina State University, Raleigh, NC, USA, 99 p.
- Bacheler, N.M., J.W. Neal, and R.L. Noble. 2004. Reproduction of a landlocked diadromous fish population: bigmouth sleeper *Gobiomorus dormitor* in a reservoir in Puerto Rico. *Caribbean Journal of Science* 40:223–231.
- Bedarf, A.T., K.R. McKaye, E.P. Van Den Berghe, L.J.L. Perez, and D.H. Secor. 2001. Initial six-year expansion of an introduced piscivorous fish in a tropical Central American lake. *Biological Invasions* 3:391–404. DOI: 10.1023/A:1015806700705
- Chapman, B.B., C. Skov, K. Hulthen, J. Brodersen, P.A. Nilsson, L.-A. Hansson, and C. Bronmark. 2012. Partial migration in fishes: definitions, methodologies and taxonomic distribution. *Journal of Fish Biology* 81:479–499. DOI: 10.1111/j.1095-8649.2012.03349.x
- Cooney, P.B. and T.J. Kwak. 2013. Spatial extent and dynamics of dam impacts on tropical island freshwater fish assemblages. *Bioscience* 63:176–190. DOI: 10.1525/bio.2013.63.3.6.
- Cronin, L.E. and A.J. Mansueti. 1971. The biology of the estuary. In: P.A. Douglas and R.H. Stroud, eds. *A Symposium on the Biological Significance of Estuaries*. Sport Fishing Institute, Washington, DC, USA, p. 14–39.
- Current Results. 2015. Annual average temperature and rainfall in Puerto Rico. www.currentresults.com/Weather/Puerto-Rico/annual-average-temperature-rainfall.php. (viewed on 10/16/2015).
- Darnell, R.M. 1962. Fishes of the Rio Tamesi and related coastal lagoons in east-central Mexico. University of Texas: Mexico Publications of the Institute of Marine Science 8:299–365.
- Enders, E.C., K.D. Clarke, C.J. Pennell, L.M. Neil Ollerhead, and D.A. Scruton. 2007. Comparison between PIT and radio telemetry to evaluate winter habitat use and activity patterns of juvenile Atlantic salmon and brown trout. *Developments in Hydrobiology* 195:231–242. DOI: 10.1007/s10750-006-0562-9
- Freund, J.G. and K.J. Hartman. 2002. Influence of depth on detection distance of low-frequency radio transmitters in the Ohio River. *North American Journal of Fisheries Management* 22:1301–1305. DOI: 10.1577-8675(2002)022<1301:IODODD>2.0.CO;2.
- Gerking, S.D. 1953. Evidence for the concepts of home range and territory in stream fishes. *Ecology* 34:347–365. DOI: 10.2307/1930901.
- Gerking, S.D. 1959. The restricted movement of fish popula-

- tions. *Biological Reviews* 34:221–242. DOI: 10.1111/j.1469–185X.1959.tb01289.x.
- Gilmore, R.G. 1992. Bigmouth sleeper, *Gobiomorus dormitor*. In: C.R. Gilbert, ed. *Rare and Endangered Biota of Florida*, Vol. 2 Fishes. University of Florida Press, Gainesville, FL, USA, p. 105–111.
- Gilmore, R.G. and P.A. Hastings. 1983. Observations on the ecology and distribution of certain tropical peripheral fishes in Florida. *Florida Scientist* 46:31–51.
- Gowan, C., M.K. Young, K.D. Fausch, and S.C. Riley. 1994. Restricted movement in resident stream salmonids: a paradigm lost? *Canadian Journal of Fisheries and Aquatic Sciences* 51:2626–2637. DOI: 10.1139/f94–262.
- Harris, N. 2007. Evaluation of hatchery spawning techniques, natural reproductive cycles, and growth rates of the bigmouth sleeper *Gobiomorus dormitor* in Puerto Rico. M.Sc. thesis. University of Arkansas at Pine Bluff, Pine Bluff, AR, USA, 72 p.
- Harris, N.J., J.W. Neal, T.D. Sink, and P.W. Perschbacher. 2012. Determination of the spawning season of bigmouth sleeper in Puerto Rico by examination of gonad maturation and reproductive hormone cycles. *Gulf and Caribbean Research* 24:41–50. DOI: 10.18785/gcr.2401.06
- Hernández-Saavedra, R., J.A. Martínez-Pérez, N.J. Brown-Peterson, and M.S. Peterson. 2004. Gonadal development and sexual dimorphism of *Gobiomorus dormitor* from the estuarine system of Tecolutla, Veracruz, Mexico. *Gulf and Caribbean Research* 16:95–100. DOI: 10.18785/gcr.1601.16
- McDowall, R.M. 2007. On amphidromy, a distinct form of diadromy in aquatic organisms. *Fish and Fisheries* 8:1–13. DOI: 10.1111/j.1467–2979.2007.00232.x
- McKaye, K.R., D.J. Weiland, and T.M. Lim. 1979. The effect of luminance upon the distribution and behavior of the Eleotrid fish *Gobiomorus dormitor*, and its prey. *Review of Canadian Biology* 38:27–36.
- Metcalf, J., G.P. Arnold, and R.M. McDowall. 2002. Migration. In: P.J.B. Hart and J.D. Reynolds, eds. *Handbook of Fish Biology and Fisheries*, Vol. 1. Blackwell Science, Oxford, UK, p. 175–199.
- Myers, G.S. 1949. Usage of anadromous, catadromous and allied terms for migratory fishes. *Copeia* 1949:89–97. DOI: 10.2307/1438482
- Neal, J.W., C.G. Lilyestrom, and T.J. Kwak. 2009. Factors influencing tropical island freshwater fishes: species, status, and management implications in Puerto Rico. *Fisheries* 34:546–554. DOI:10.1577/1548–8446–34.11.546
- Neal, J.W., C.M. Adelsberger, and S.E. Lochmann. 2011. A comparison of larval fish sampling methods for tropical streams. *Marine and Coastal Fisheries: Dynamics, Management, and Ecosystem Science* 4:23–29. DOI: 10.1080/19425120.2011.651521.
- Nickum, J.G., H.L. Bart, P.R. Bowser, I.E. Greer, C. Hubbs, J.A. Jenkins, J.R. MacMillan, J.W. Rachlin, J.D. Rose, P.W. Sorensen, and J.R. Tomasso. 2004. *Guidelines for the Use of Fishes in Research*. American Fisheries Society, Bethesda, Maryland, USA, 57 p.
- Nordlie, F.G. 1979. Niche specificities of Eleotrid fishes in a tropical estuary. *International Journal of Tropical Biology and Conservation* 27:35–50.
- Nordlie, F.G. 1981. Feeding and reproductive biology of Eleotrid fishes in a tropical estuary. *Journal of Fish Biology* 18:97–110. DOI: 10.1111/j.1095–8649.1981.tb03764.x.
- Peters, L.M., U.G. Reinhardt, and M.A. Pegg. 2008. Factors influencing radio wave transmission and reception: use of radiotelemetry in large river systems. *North American Journal of Fisheries Management* 28:301–307. DOI: 10.1577/M06–146.1
- SAS Institute. 1990. *SAS/STAT User's Guide*, Version 6, 4th ed. Cary, NC, USA, 1848 p.
- Secor, D.H. and L.A. Kerr. 2009. Lexicon of life cycle diversity in diadromous and other fishes. *American Fisheries Society Symposium* 69:537–556.
- Smith, W.E. and T.J. Kwak. 2014. Otolith microchemistry of tropical diadromous fishes: spatial and migratory dynamics. *Journal of Fish Biology* 84:913–928. DOI: 10.1111/jfb.12317.
- Winemiller, K.O. and B.J. Ponwith. 1998. Comparative ecology of Eleotrid fishes in Central American coastal streams. *Environmental Biology of Fishes* 53:373–384. DOI: 10.1023/A:1007422821071.
- Whitfield, A.K. 2005. Preliminary documentation and assessment of fish diversity in sub Saharan African estuaries. *African Journal of Marine Science* 27:307–324. DOI: 10.2989/18142320509504089.
- Winter, J. 1996. Advances in underwater biotelemetry. In: B.R. Murphy and D.W. Willis, eds. *Fisheries Techniques*, 2nd ed. American Fisheries Society, Bethesda, MD, USA, p. 555–590.

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***Rangia cuneata* clam decline in Lake Pontchartrain from 2001 to 2014 due to an El Niño Southern Oscillation shift coupled with a period of high hurricane intensity and frequency**

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RANGIA CUNEATA CLAM DECLINE IN LAKE PONTCHARTRAIN FROM 2001 TO 2014 DUE TO AN EL NIÑO SOUTHERN OSCILLATION SHIFT COUPLED WITH A PERIOD OF HIGH HURRICANE INTENSITY AND FREQUENCY

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ABSTRACT: The clam, *Rangia cuneata*, occurs in low salinity zones of Atlantic and Gulf of Mexico estuaries. Prior to 2001, it was abundant in Lake Pontchartrain, a large and shallow oligohaline estuary, located north of New Orleans. However, density abruptly decreased by 96% after an extreme drought from an El Niño Southern Oscillation (ENSO) shift (1998–2000). Stable higher salinities resulted in the establishment of higher salinity communities. The hooked mussel, *Ischadium recurvum*, increased abruptly and colonized live *R. cuneata*. Competition from *I. recurvum* and the *R. cuneata* decline occurred at the end of the drought, despite a decrease in salinity. After the drought, there was a period of high hurricane intensity and frequency which stifled clam recovery. Their densities remained relatively low due to declines from hurricane disturbances in 2005, 2008 and 2012. In 2014, after two years without severe hurricane effects, density and biomass recovered to baseline years (1954 and 1997). Barrier island erosion caused by Hurricane Georges in 1998 and adverse effects of the ENSO drought on wetlands may have set the stage for increased effects of subsequent hurricanes. High relative sea level rise, wetland loss and erosion of barrier islands, which occurred with each successive storm, potentially increased the size of surges. Surges resuspended sediments which buried clams and abruptly increased salinity and lowered dissolved oxygen. Salinity stratification persisted after storms and caused hypoxia in bottom waters. Storm surge interactions with discharges from the Bonnet Carré Spillway, a Mississippi River flood diversion structure, and the MRGO, a ship channel, also exacerbated hurricane effects.

KEY WORDS: drought, storm surges, sea level rise, climate change, *Ischadium*

INTRODUCTION

Rangia cuneata (Sowerby I, 1832) is indigenous to low salinity areas of Atlantic and Gulf of Mexico estuaries, and has recently been reported from Europe (Verween et al. 2006). It is an important component of estuarine ecosystems because it provides food for fishes, crabs, shrimp, waterfowl, and humans (LaSalle and de la Cruz 1985), and performs important environmental services, including water filtration and shell production, which stabilizes soft bottoms. It generally occurs at salinities < 18, but is more abundant in oligohaline (0.5–5) waters where it experiences less competition and predation due to low species diversity (LaSalle and de la Cruz 1985). Fairbanks (1963) studied the growth rate of *R. cuneata* in 2 areas of eastern Lake Pontchartrain during 1957 and 1958. His estimated annual growth in length for 3 years were 15–20 mm (year 1), 20–29 mm (year 2) and 24–34 mm (year 3). LaSalle and de la Cruz (1985) estimated the average life span to be 4–5 years and, based on several studies reported in Louisiana, spawning may be continuous with most spawning from March to May and late summer to November. Spawning, however, has been reported by Cain (1975) to be dependent upon a salinity shift of ± 5 at temperatures above 15°C. *Rangia cuneata* has been considered an indicator species in Lake Pontchartrain, Louisiana, where its density (N/m²) has been proposed as a measure of recovery from shell dredging, saltwater intrusion (Abadie and Poirrier 2000), and general anthropogenic disturbances (Houck 1989).

The earliest baseline data for *R. cuneata* from Lake Pontchartrain were obtained by Suttkus et al. (1954). They only studied clams > 20 mm and reported a mean density of 95/m². Studies conducted in 1969–1972 (Tarver and Dugas 1973), 1978–1980 (Sikora and Sikora 1982) and 1982–1983 (Poirrier et al. 1984) indicated declines in large clams (> 20 mm) from 1954 baseline densities, which were mainly attributed to commercial shell dredging (USACE 1987). Densities increased to 1954 levels after dredging was stopped in 1990, but large clams were still absent from a 250 km² area subject to saltwater intrusion from the MRGO (Abadie and Poirrier 2000). The post-dredging increase in large clam density from 1996 through 2000 was regarded as a return to normal conditions. However, there was an unexpected, abrupt 96% density decrease between 2000 and 2001 (Figure 1) after a severe drought (Visser et al. 2002), linked to an El Niño Southern Oscillation (ENSO) shift from the strongest El Niño in the last century (Zheng et al. 2003) to a La Niña from 1997 to 2001 (Cho and Poirrier 2005, Poirrier et al. 2009). We were uncertain about the cause of the 2001 clam density crash after the drought and whether or not recovery to pre-drought density levels would eventually occur. Densities have remained substantially lower than pre-drought levels (1998–2000) even after the closure of the MRGO (Poirrier 2013). Poirrier et al. (2013) proposed that clam density recovery from the 2001 density crash did not occur because of storm surges from a period of high

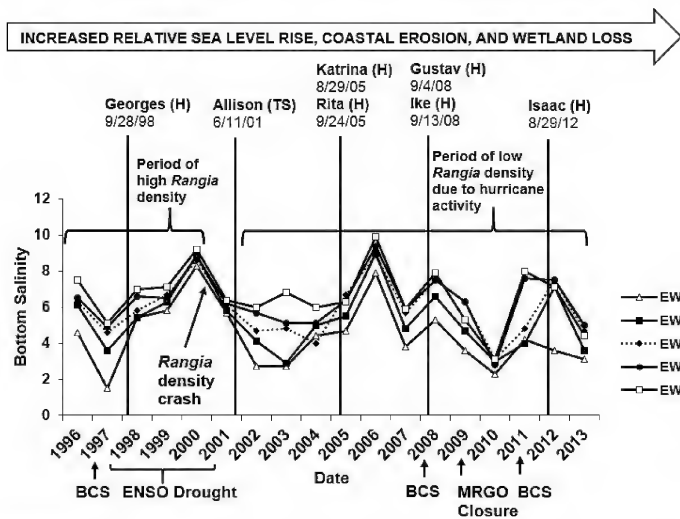


Figure 1. Pre-drought period of high clam density, the post-drought 2001 density crash, and period of low density associated with hurricane activity. Years sampled during 1996–2013 and events including: hurricanes and tropical storms, Bonnet Carré Spillway openings (BCS), El Niño Southern Oscillation (ENSO) drought years, Mississippi River Gulf Outlet (MRGO) closure, spatial and temporal bottom salinity changes across the width of the Lake on 5 E–W transects (EW1 – EW5). H = hurricanes and TS = tropical storms.

hurricane intensity and frequency (Figure 1). These surges were enhanced by relative sea-level rise (Burkett 2008, Roth 2010), associated accelerated wetland and barrier island loss (Penland et al. 2001), and a possible increase in the cross-sectional area of the tidal passes (Li et al. 2009) during and after severe hurricanes. Anthropogenic climate change may have also directly increased hurricane intensity (Burkett 2008). Severe hurricane surges can have a direct lethal effect on benthic invertebrates (Poirrier et al. 2008): they disturb sediments, which bury clams, abruptly increase salinity, and produce water column stratification with anoxic and hypoxic bottom water (Poirrier et al. 2013).

Unlike most estuaries, Lake Pontchartrain has a slow flushing rate due to narrow tidal passes and weak lunar tides. This makes it more sensitive to variation in rainfall and stratification from storm surges. High sediment re-suspension from wind occurs due to its shallow depth and large fetch. Re-suspended sediments are not flushed, but rather are re-deposited on the bottom (Flowers and Isphording 1990). Earlier studies based only on large clams (Poirrier et al. 2009, Poirrier et al. 2013) documented the significant, potentially drought-driven 2000–2001 population crash, but did not find recovery. They did not have the statistical power to determine significant annual differences due to hurricanes because large clam densities were low and there was high variance due to only 5 sampling sites being used.

This study was divided into two components, the La Niña drought and hurricane effects. The overall goal of the drought study was to investigate the specific causes of the 2001 crash by examining 1) changes in the density of 5 mm

size classes and total biomass, 2) the relationship between density and salinity changes, 3) competition from substantial increases in the hooked mussel, *Ischadium recurvum*, and other related changes in community structure of *R. cuneata* from 1998 through 2005. Goals of the hurricane effects and recovery study were 1) to determine if there were significant reductions in clam density and biomass of major size classes (15 mm) in years after major hurricanes compared to non-hurricane years, and 2) to determine if recovery of *R. cuneata* to 1954 and 1997 baseline levels occurred after 2013 and 2014, two years without hurricanes or high-salinity related stress. This study was part of a general, long-term study of benthic invertebrate community structure, which supported our results, but was too complex to include with this *R. cuneata* study.

MATERIALS AND METHODS

Area of Study

Lake Pontchartrain is a large, shallow, oligohaline estuary located north of New Orleans, Louisiana, USA (Figure 2). Its two relatively narrow tidal passes open into Lake Borgne, the Biloxi Marsh and the Chandeleur Islands, which are part of a deteriorating Mississippi River deltaic system abandoned by the Mississippi River (Saucier 1963). Lake Pontchartrain has a mean salinity of 3.9 (Francis et al. 1994), a surface area of 1,630 km², and a mean depth of 3.7 m (Sikora and Kjerfve 1985). Higher salinity waters enter from the east through these tidal passes. It receives fresh water from rivers and bayous to the west and north, urban outfall canals on the south shore, and from periodic openings and leakage of the Bonnet Carré Spillway, a Mississippi River flood diversion structure (Brammer et al. 2007). Therefore, salinity generally decreases from east to west (Swenson 1980). It is adversely affected by urban and agricultural runoff, shoreline modification (Penland et al. 2001), Bonnet Carré Spillway openings, and was affected by saltwater intrusion from a former ship channel, the Mississippi River Gulf Outlet (MRGO), that produced salinity stratification and periodic bottom water anoxia/hypoxia from 1968 to 2009 (Poirrier 2013).

Field and Laboratory

In the field, 3 replicate infaunal samples were taken at sites sampled throughout Lake Pontchartrain (Figure 2) using a 15 cm² petite Ponar dredge. Samples were separated from fine sediment using a Wildco 12 L, 0.6 mm sieve bucket, and then preserved in 10% Borax-buffered formalin with Rose Bengal stain. Salinity and dissolved oxygen were measured at 30 cm below the surface and above the bottom with a YSI-85 S–C–T meter. In the laboratory, samples were rinsed in a 0.5 mm sieve and preserved in 70% isopropanol. Clams were removed from rinsed dredge samples, placed in 5 mm size classes and counted. Clams < 5 mm were excluded from the data set due to their low numbers

when large clams were present, their rapid and high recruitment after various stressful events, and difficulty in separating small *R. cuneata* from small dwarf surf clams, *Mulinia lateralis*. Number (N) of individuals sampled by the dredge was converted to N/m^2 . *Rangia cuneata* dry weight biomass was determined by drying soft tissue at 60°C for 48 hours (Edmondson 1971). Soft tissue dry weight was regressed against shell length to obtain the predictive equation used in this study (Spalding et al. 2007). *Ischadium recurvum*, whose population exploded during the drought, was also counted and placed in 5 mm size classes.

In the analysis of samples, possible effects of the La Niña drought during 1998–spring 2005, and hurricanes during 2004–2014 were treated separately because only samples from the E–W and N–S transects with 7 sites and 3 replicates per site (Figure 2) were taken during the drought study period. April 2005 was used in the drought analysis instead of fall 2005 samples, because fall samples were severely affected by Hurricane Katrina. Samples from all of Lake Pontchartrain during the period of high hurricane frequency were from 30 sites (Figure 2) with 3 replicates per site. Lake Pontchartrain fall 1997 data were obtained during a strong El Niño year, which was not affected by droughts or damaging hurricanes; these data were used in the hurricane effects analysis as a baseline year to determine recovery.

During the drought study, 7 sites on the E–W and N–S transects were sometimes sampled several times. Only 2 northern sites on the N–S transect (Figure 2, NS 1 and 2) were used because others were affected by saltwater intrusion and hypoxia from a navigation canal. Year and month sampled and total number of samples (N) are as follows: 1998 November (21); 1999 April, August, and October (63); 2000 January, May, and July (63); 2001 April (21); 2002 August (21); 2003 November (21); 2004

April, October, and November (62); and 2005 April (21). We combined monthly within-year data because an early analysis using uncombined monthly data excluding < 5 mm clams indicated no significant differences from monthly within-year data, and past analyses using separate monthly samples gave similar results (Poirrier et al. 2013). This was due to spawning in the sub-tropical waters of Lake Pontchartrain being almost continuous and dependent upon random salinity shifts rather than seasonal changes. The only seasonal sample that could have affected recruitment was January 2000, which was included in the year before the 2001 crash. Also, the crash was so severe that decline and recovery of size classes was easy to detect.

In the hurricane study, sites from the all of Lake Pontchartrain were sampled, and all were obtained during the fall (October and November) with a total of 90 per year (Figure 2). Figure 1 presents time sequences for the drought, hurricane events, and other disturbances.

Statistical Analyses

One-way analysis of variance (ANOVA) was used to test for differences among annual clam densities. Clam densities were \log_{10} transformed to produce a data distribution acceptable for parametric analyses. Although most data exhibited mild departures from normality, this was considered acceptable because the ANOVA is considered robust with regard to minor departures from normality (Gotelli and Ellison 2004, Neter et al. 1990). *Post-hoc* Tukey pairwise comparisons were used to test for differences among means. Analyses were performed with SPSS (IBM SPSS Statistics for Windows, Version 21.0. Armonk, NY) using a 0.05 significance level.

For the drought analysis, ANOVA was used to determine significant annual changes in *R. cuneata* density, size classes and biomass, coupled with significant differences in salinity, and *I. recurvum* density. For the hurricane disturbance effects (the combination of clam burial, abrupt salinity change with anoxia and hypoxia) and recovery analysis, ANOVA was used to determine annual changes in *R. cuneata* density, biomass, and salinity. An ANOVA was also run on clams > 20 mm with data from 1954, 1997, 2004, 2005, 2006, 2008, 2012, 2013, and 2014 to see if recovery had occurred based on the earliest 1954 > 20 mm clam densities. A one-way between-groups multivariate analysis of variance (MANOVA) was performed to investigate differences in *R. cuneata* size classes during the drought period. Necessary Bonferroni alpha level adjustments were made in the MANOVA. Dependent variables were the clam densities of 5 mm size classes (6–10, 11–15, 16–20, 21–25, 26–30, 31–35, and 36–40). The independent variable was year.

RESULTS

La Niña Drought

There were significant differences in clam densities (all

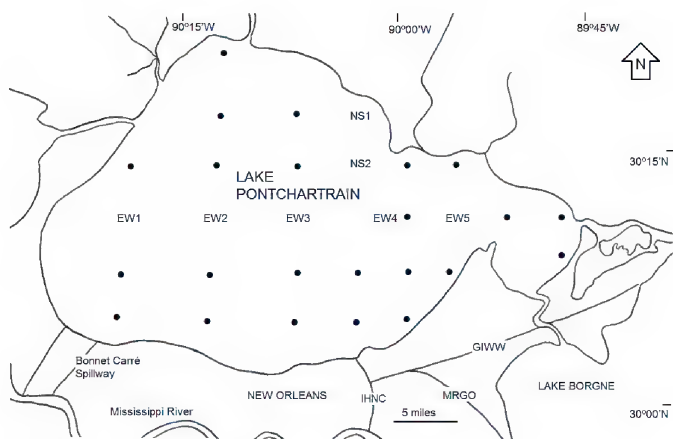


Figure 2. *Rangia cuneata* sampling sites in Lake Pontchartrain, indicating sites on the E–W and N–S transects and other Lake Pontchartrain sites (solid circles ●). IHNC = Inner Harbor Navigational Canal, MRGO = Mississippi River Gulf Outlet, and GIWW = Gulf Intracoastal Waterway.

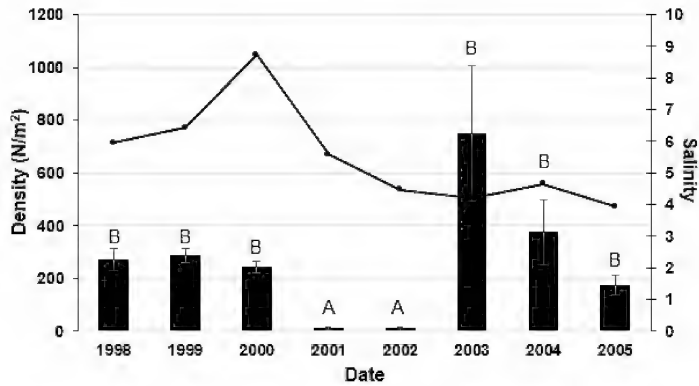
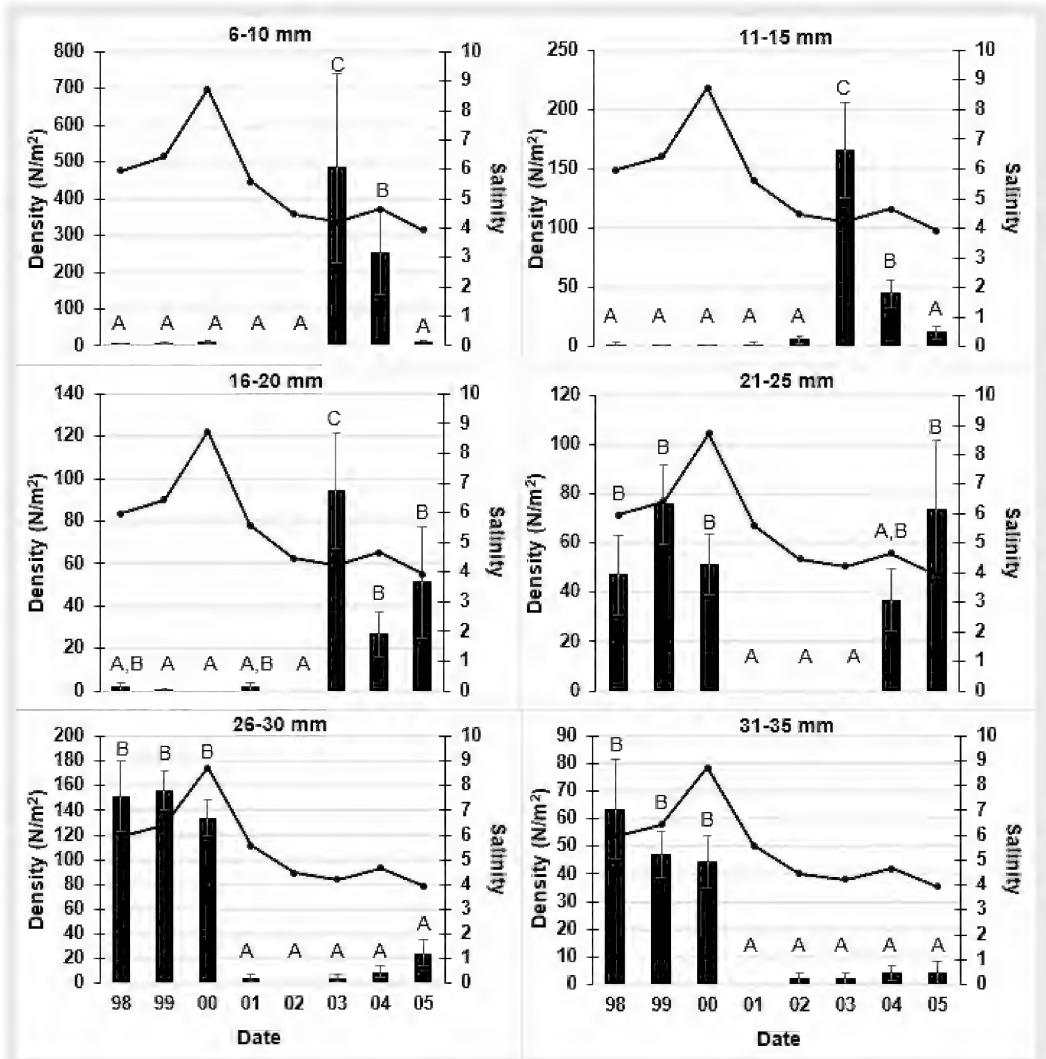


Figure 3. Annual mean (\pm se) *Rangia cuneata* clam densities (6–55 mm/m²) before (1998), during (1999, 2000 and early 2001), and after (late 2001–2005) the La Niña drought years. Dots on the black line indicate annual mean bottom salinity from samples at the time of collection. Groups with the same letters indicate no significant difference (Tukey post-hoc test, $p < 0.05$).

clams 6–55 mm) among years (1998–2005; $F_{(7,265)} = 16.98$; $p < 0.001$) (Figure 3). The Tukey comparisons, with $p < 0.001$ for all results, indicated a significant decline in total clam density in 2001 and 2002, and density returned to 1998–2000 levels in 2003–2005 (Figure 3). Between July 2000 and April 2001, density was reduced to only 10 clams/m², a 96% reduction. MANOVA found a statistically significant difference in density among years on the combined size classes ($F_{(49,1319)} = 8.69$; $p < 0.001$) (Figure 4). When the results of the dependent variables were considered separately, using a Bonferroni adjusted alpha level of 0.01, only the 36–40 mm size class did not show significant annual differences because of their low density.

Figure 4. Plots of annual mean (\pm se) *Rangia cuneata* densities (N/m²) in size classes: 6–10, 11–15, 16–20, 21–25, 26–30 and 31–35 mm, set against years before (1998), during (1999, 2000 and early 2001), and after (late 2001–2005) the La Niña drought. Dots on the black line indicate annual mean bottom salinity from samples at the time of collection. Groups with the same letters indicate no significant difference (Tukey post-hoc test, $p < 0.05$).



Changes in density occurred in all size classes (Figure 4). When annual densities within individual size classes were compared, smaller clams (6–20 mm) were not very dense prior to the 2001 population crash, when large clams were dense (Figure 4). However, they rapidly increased in 2003 and then declined as larger size classes became established. In contrast, larger clams (21–35 mm) were more dense prior to the crash and, depending on size, slowly recovered. By 2005 there was a general trend toward return to pre-crash density levels for the 21–25 mm clams, but not the largest clam class (26–35 mm; Figure 4).

The ANOVA analysis of total biomass of 6–55 mm clams from 1998–2005 found significant differences among years ($F_{(7,265)} = 19.9$; $p < 0.001$) (Figure 5). The Tukey comparisons support a significant decrease in biomass from 2000 to 2001 (95%), and an increase in 2003–2005, but no recovery to 1998–2000 levels. Density and biomass results differed across years because the few but large clams that occurred from 1998 through 2000 had a high total biomass, whereas the few clams of any size had relatively low biomass during 2001 and 2002; in 2003 dense small clams with low biomass became established while in 2004 and 2005 there was

Figure 5. Annual mean (\pm se) dry weight biomass (g/m^2) of *Rangia cuneata* before (1998), during (1999, 2000 and early 2001), and after (late 2001–2005) the La Niña drought years. Dots on the black line indicate annual mean bottom salinity from samples at the time of collection. Groups with the same letters indicate no significant difference (Tukey post-hoc test, $p < 0.05$).

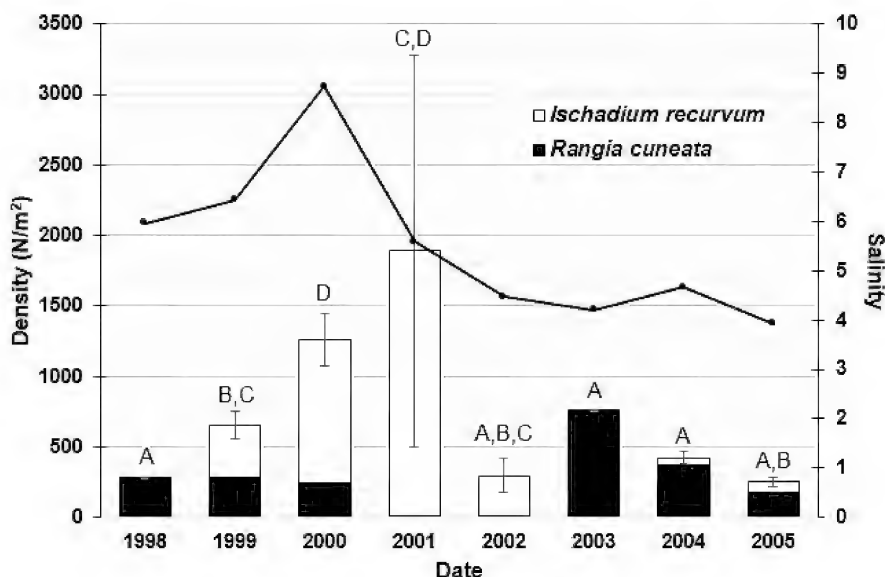
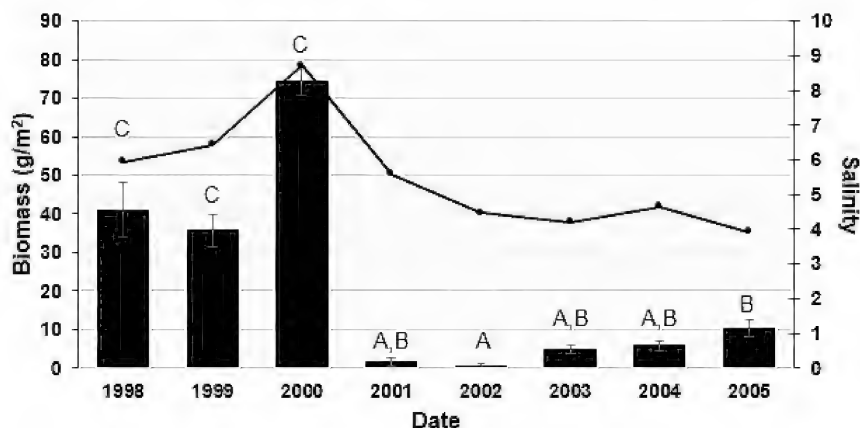


Figure 6. Annual *Rangia cuneata* and *Ischadium recurvum* mean (\pm se) densities of *Ischadium recurvum* before (1998), during (1999, 2000, and early 2001), and after (late 2001–2005) the La Niña drought years. Dots on the black line indicate annual mean bottom salinity from samples at the time of collection. Groups with the same letters indicate no significant difference in *Ischadium recurvum* densities (Tukey post-hoc test, $p < 0.05$).

a shift to fewer larger clams with higher biomass (Figure 5). A MANOVA analysis of annual changes among 5 mm size classes found significant results ($F_{(49, 1319)} = 6.73$; $p < 0.001$) and the Tukey comparisons agreed with groupings found in the size class density analysis.

The *R. cuneata* decline occurred within a 10 month period between July 2000 and April 2001 after the drought-driven peak in salinity values (Figures 3 and 5). Salinity increased prior to the decline, but the decline occurred after the drought when the salinity decreased. This indicates that salinity was not the direct cause of the density crash. Salinity increased to 8.8 in 2000 and decreased to 6.0 in April 2001, and was further reduced to 4.6 by Tropical Storm Allison in June 2001. Based on an ANOVA, the 2000 increase in salinity was significantly different ($F_{(7,265)} = 70.1$; $p < 0.001$) compared to other years from 1998 through 2005.

ANOVA and Tukey comparisons found significant changes in *I. recurvum* density from 1998–2005 (Figure 6). *Rangia cuneata* decreased in 2001 after the 1999 and 2000 *I. recurvum* increase. Prior to 1999, few ($< 20/\text{m}^2$) small (< 5 mm) *I. recurvum* occurred mainly at site E–W 5 at higher salinities in the eastern end of Lake Pontchartrain (Figure 2). As salin-

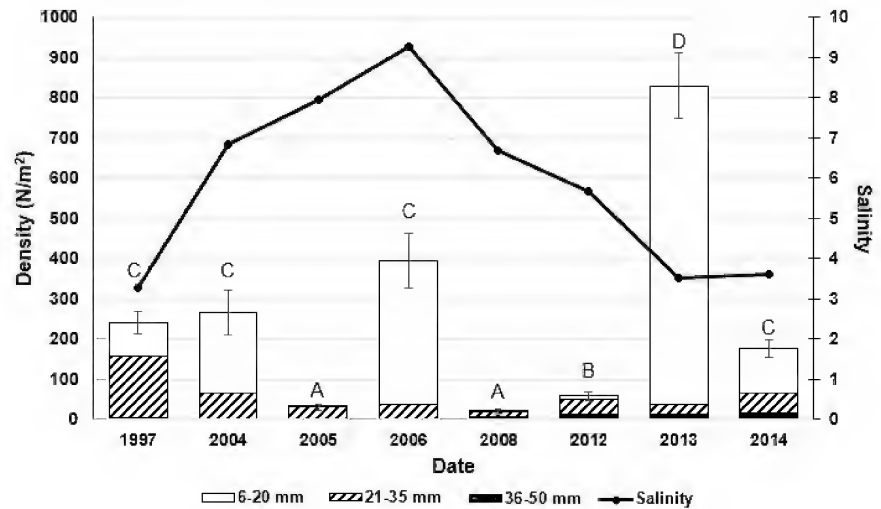
ity increased at all sites from 1999 through 2000, *I. recurvum* colonized all E–W sites (Figure 2) and mussels up to 35 mm became abundant (Figure 6). This rapid increase in larger mussels on live *R. cuneata* shells often formed small spherical reefs about 30 cm in diameter. Other filter feeders including the barnacles, *Balanus improvisus*, and *B. eburneus*, Conrad's false mussel, *Mytilopsis leucophaeta*, and the bryozoan, *Conopium* sp., became abundant on *I. recurvum* in these reefs. These specialized filter feeders, in addition to *I. recurvum*, probably removed a broad range of particles that were normally available to *R. cuneata*. After 2002, *I. recurvum* abruptly decreased and *R. cuneata* increased (Figure 6).

Hurricane Effects

The ANOVA analysis comparing annual mean densities of 6–50 mm clams for years 1997, 2004, 2005, 2006, 2008, 2012, 2013, and 2014 found significant ($F_{(7,712)} = 50.56$; $p < 0.001$) differences among years (Figure 7). Tukey comparisons (Figure 7) revealed that hurricane years 2005 (Katrina and Rita), 2008 (Ike and Gustav), and 2012 (Isaac) differed from all other years ($p < 0.001$).

A MANOVA of annual differences among 15 mm size

FIGURE 7. Annual mean density of 6–50 mm *Rangia cuneata* clams/m² from fall samples of 30 representative sites throughout Lake Pontchartrain during 1997, 2004, 2005 (Katrina), 2006, 2008 (Ike & Gustav), 2012 (Isaac), 2013 and 2014. Representation of 15 mm size classes in stacked columns is presented with standard errors for total density. Dots on the black line indicate annual mean bottom salinity from samples at the time of collection. Groups with the same letters indicate no significant difference (Tukey post-hoc test, $p < 0.05$).



classes found significant differences ($F_{(21, 2246)} = 29.54$; $p < 0.001$). Tukey comparisons among years for 6–20 mm clams clearly separated the hurricane years from all others and separated 2013 from all years. There were abrupt decreases of 6–20 mm clams in hurricane years and rapid increases when followed by years without hurricanes or salinity stress, such as 2013. 2014 was different from all other years in the 21–35 mm comparison with poor separation of other years. In the 36–50 mm comparison, years 2005 through 2014 grouped together, but 2005 and 2014 also grouped with 1997 and 2004 (Figure 7).

Biomass results differed from density results due to biomass increasing exponentially with clam size. The ANOVA of annual mean biomass of 6–50 mm clams for years 1997, 2004, 2005, 2006, 2008, 2012, 2013, and 2014 found significant differences ($F_{(7, 712)} = 9.43$; $p < 0.001$) (Figure 8). Tukey comparisons showed 2004, 2005, 2006, 2008 and 2012 were grouped into a common subset (Figure 8). This group contains three hurricane years, 2005, 2008 and 2012, but also contains 2004, a year that was still recovering from the La Niña drought, and 2006, a year after the acute disturbances caused by Katrina and Rita in 2005. Within this group, 2004, 2006 and 2012 also group with 2013 and 2014, two years with relatively low salinity and without severe damaging hurricanes. However, 2013 and 2014 also grouped with 1997, indicating overall biomass recovery to 1997 levels in 2013 and 2014. Tukey comparisons of the three 15 mm size classes based on biomass did not give distinct groups except for 2013 differing from all other years for the 6–20 mm clams, and 1997 differing from all other years for 21–35 mm clams. The 36–50 mm comparisons did not give clear separations of groups among years. However, groups supported differences between 2012 through 2014 compared to 2004, 2005 and 2006, which supports the increase in biomass due to the increase in large clams during and after 2012.

Large Clam (> 20 mm) changes over time

An ANOVA of samples of large clams (> 20 mm) from years with a minimum of 30 sites from 1954 data (the earliest baseline data from Suttikus et al. (1954)) through 2014 showed significant differences among years ($F_{(8, 744)} = 15.72$; $p < 0.001$). The Tukey comparisons grouped 2004 and 2014 with 1954 indicating recovery to 1954 levels, but not 1997 levels, based on > 20 mm clams.

DISCUSSION

General Overview

Study results using more sites and size classes than Poirrier et al. (2013) appear to confirm that the La Niña drought: (1) contributed to the abrupt 2001 population crash of *R. cuneata*, (2) appeared to affect all size classes and biomass, (3) resulted in a shift to a higher salinity community that caused *I. recurvum* overgrowth, and (4) decreased large clam density and biomass which did not completely recover prior to Hurricane Katrina. Links between the 1997–2001 ENSO shift and low precipitation with increased salinity were not included in this study because they were established in earlier studies of submersed aquatic vegetation in Lake Pontchartrain (Cho and Poirrier 2005). Based on comparisons of hurricane and non-hurricane years, we found that hurricanes Katrina, Ike, Gustav and Isaac reduced clam densities and total biomass, and recovery occurred by 2014, during relatively low salinities and after 2 years without hurricanes.

La Niña Drought

Rangia cuneata is an estuarine endemic (Carriker 1967). Although adults are physiologically capable of tolerating limnetic to polyhaline waters, it is more abundant in the oligohaline zone where it is protected from predation and competition due to low species richness (Remane and Schlieper 1971). In nature, *R. cuneata* occurs at salinities up to 18 (LaSalle and de la Cruz 1985) and it is known to survive at

a salinity of 32 in the laboratory (Bedford and Anderson 1972). An important aspect of the drought was a change in the salinity regime that resulted in persistent, relatively high salinity throughout the estuary with no episodes of low salinity from runoff (Poirrier et al. 2009). Periodic low salinity, characteristic of poikilohaline estuaries, keep higher salinity species, including competitors and predators, from becoming established (Boesch 1977). During the drought, stable salinity allowed the establishment of higher salinity communities, which potentially increased biotic stress on the *R. cuneata* population due to increased competition and predation. We obtained reasonable results in the drought analysis, despite the time of sampling varying among years, because *R. cuneata* spawning is almost continuous in Lake Pontchartrain and stimulated by salinity shifts rather than seasonal changes. The lack of salinity shifts required for *R. cuneata* spawning and recruitment (Cain 1975) during the drought may have directly contributed to the decline, but was probably not the primary cause because density levels of 6–10 and 11–15 mm clams remained the same from 1998 through 2002.

The *I. recurvum* increase characterized the community shift and also appeared to be another important causal factor for the *R. cuneata* decline. *Ischadium recurvum* is more characteristic of a euryhaline marine species, does not survive episodes of fresh water, and is more abundant in the mesohaline zone. It can be invasive and rapidly colonizes the Eastern oyster, *Crassostrea virginica*, and the fouling that results interferes with commercial oyster production (Coleman 2003). Although some oysters colonized artificial fishing reefs in the southwestern part of Lake Pontchartrain, we did not find any oyster spat on *R. cuneata*. However, without the *I. recurvum* invasions, we speculate that oyster

colonization on *R. cuneata* may have occurred.

Rangia cuneata densities did not change from 1998 through 2000. Although clams were subjected to increasing stress from competition in 1999 and 2000, the crash did not occur until 2001. It occurred under conditions of decreased salinity, but while a higher salinity community was still present. This time lag supports *I. recurvum* overgrowth rather than the singular effect of salinity as the cause of the crash. A similar explosion in *I. recurvum* abundance occurred on the New Orleans seawall during the same time period (Poirrier, unpublished data). *Ischadium recurvum* was absent in 1998, dominated the epifaunal community, reached a maximum size of 60 mm in 2001, and was absent in 2003. Lasting effects of low recruitment of *R. cuneata* during the drought and subsequent low-levels of reproduction and future recruitment appeared to interact synergistically with *I. recurvum* overgrowth in causing the decline.

Increased competition and predation from numerous specialized species in the higher salinity communities could have also negatively affected most aspects of *R. cuneata*'s life cycle (e.g., larval predation by jellyfish, competition from other specialized filter feeders). However, increased predation on *R. cuneata* clams by bivalve fish predators does not appear to be the major cause of the decline because the *I. recurvum* population, which should have also been subjected to predation, exploded while the *R. cuneata* population crashed.

ENSO shifts to La Niña that occurred at other times (1973–1976; 1988–1989) may have affected clam abundance. However, no La Niña events occurred during periods of low *R. cuneata* densities attributed to shell dredging (Tarver and Dugas 1973, Sikora and Sikora 1982, Poirrier et al. 1984), suggesting that these low density periods should

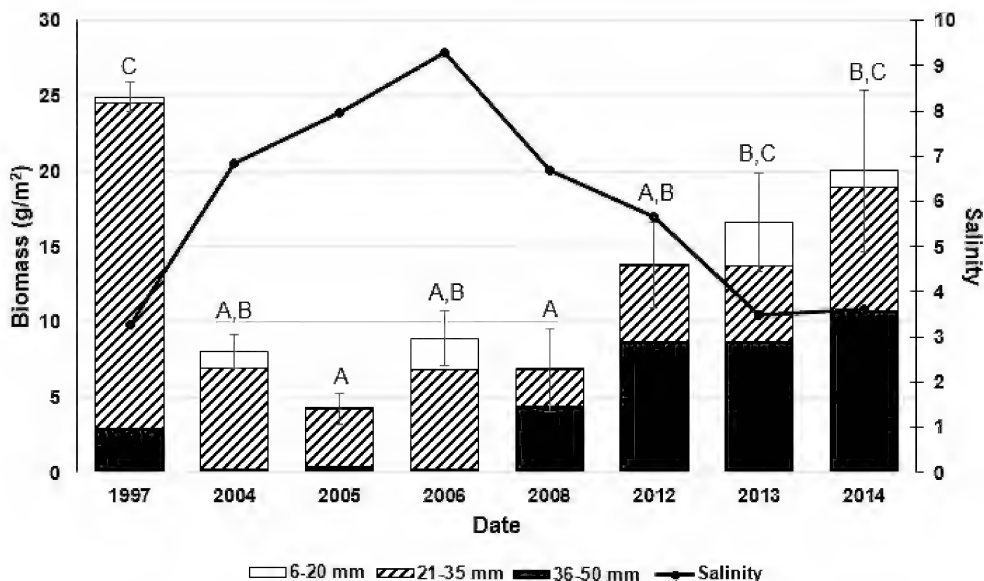


Figure 8. Annual mean dry weight biomass (g/m^2) of *Rangia cuneata* clams from fall samples of 30 representative sites throughout Lake Pontchartrain during 1997, 2004, 2005 (Katrina), 2006, 2008 (Ike & Gustav), 2012 (Isaac), 2013 and 2014. Representation of 15 mm size classes in stacked columns is presented with standard errors for total biomass. Dots on the black line indicate annual mean bottom salinity from samples at the time of collection. Groups with the same letters indicate no significant difference (Tukey post-hoc test, $p < 0.05$).

not be attributed to ENSO shifts.

Hurricane Effects and Recovery

The lack of complete recovery of large size class clams and overall biomass after the drought appears to have contributed to the later damage to *R. cuneata* populations caused by hurricanes. The decrease in clam density during the hurricane years, 2005, 2008 and 2012 indicated distinct hurricane effects. These effects are supported by previous studies of benthic invertebrate impacts from Katrina and Rita (Poirrier et al. 2008, Engle et al. 2009) and Ike and Gustav (Ray 2009). We found that density changes in 6–20 mm clams accounted for most of the differences. There were significant decreases of *R. cuneata* in hurricane years and significant increases in years without hurricanes. The density reduction in hurricane years was probably due to small clams being more sensitive to direct physical hurricane damage, while their rapid recovery is due to recruitment and growth to 20 mm within a year. Possible effects of Hurricanes Isidore and Lili were not detected because they occurred in 2002 when *R. cuneata* populations were already very low due to the drought.

Annual biomass changes agreed with the density changes when the exponential relationship between biomass and size was considered. Many small clams made minor contributions to total biomass; whereas, fewer large clams made major contributions. There was a time lag in the recovery of large clams. Clams from 6 to 20 mm recovered within a year, but recovery of large clams took several years. This resulted in biomass recovery being a slow process compared to density recovery. Our 1997 baseline year had a high proportion of 21–35 mm clams when compared to later years. Large clams were almost eliminated during the 2001 crash from the drought and were slow to recover, presumably due to hurricanes in 2005 and 2008. This contributed to the low biomass prior to 2012. The steady increase in biomass from 2012 to 2014, despite the effects of Isaac, may have been due to large clams being more resistant to hurricanes, Isaac having less of an impact than earlier storms, and the closure of the MRGO in 2009 which reduced storm surges and stopped episodes of detrimental dissolved oxygen. Prior to the MRGO closure, these episodes were frequent in the area north of the Inner Harbor Navigational Canal (IHNC), but also spread to other areas of Lake Pontchartrain during late summer (Poirrier 1978, 2013). Biomass reached 1997 baseline recovery levels by 2014, following 2 years without salinity or hurricane stress. Recovery based on density occurred in 2013 and 2014, but densities statistically equal to 1997 also occurred in 2004 and 2006. Because of the rapid change in small clams less than a year old, biomass is a better indicator of recovery in a given year. However, loss of the 6–20 mm size class results in the lack of production of larger size classes in subsequent years.

Hurricane effects differ based on their track and the

landfall coastal morphology (Chen et al. 2008). Lake Pontchartrain has a higher risk of surges due to the extension of the Mississippi River delta into the Gulf which can intensify surges that form and enter from the east. A severe disturbance should be expected when a 3.7–4.9 m storm surge, such as the one that occurred during Katrina, enters a soft-bottomed estuary with a mean depth of 3.7 m. Damage and burial from storm surges were important factors in *R. cuneata* clam decline. Poirrier et al. (2008) found recently dead and dying *R. cuneata* after Katrina. After Isaac, articulated shells with non-weathered periostraca from dead *R. cuneata* clams were common in sediment samples (Poirrier et al. 2013). Shells did not show any signs of damage from predation and were apparently produced by clams dying in the sediment. Flocks et al. (2009) found that the top 1 m of sediments are constantly being reworked by major storms and circulation gyres. Because of slow flushing rates, sediments suspended by storms are not transported out of Lake Pontchartrain, but rapidly settle to the bottom across broad areas (Flowers and Isphording 1990).

O'Connell et al. (2013) found minor compositional changes in post-Hurricane Katrina and Rita fish assemblages. Individually, there were increases in *Ariopsis felis* (Hardhead Catfish) and declines in *Gobiosoma bosc* (Naked Goby), *Syngnathus scovelli* (Gulf Pipefish), *Membras martinica* (Rough Silverside), *Micropogonias undulates* (Atlantic Croaker), *Scomberomorus maculatus* (Spanish Mackerel), and *Cynoscion arenarius* (Sand Seatrout), even though some of these species were not major components of assemblages. Although clam loss may appear to have a negative effect on consumers, physical damage from hurricanes and hypoxia may make large clams more susceptible to crab and fish predators after storm events. Also, loss of large clams is typically followed by rapid recruitment of small clams which are more easily consumed by shrimp, ducks, crabs and fishes, potentially increasing secondary productivity. However, reduction in clam biomass due to the loss of large clams may result in loss of filtration and other environmental services, likely contributing to poorer water quality in nearshore habitats where adults of *G. bosc*, *S. scovelli*, and *M. martinica* reside. Additionally, higher turbidity can result in loss of SAV and the services it provides (e.g., protection from predation for *S. scovelli*, egg habitat for *M. martinica*; Spalding et al. 2007, O'Connell et al. 2013).

Little information is available about bottom water quality in Lake Pontchartrain during and immediately after hurricanes. However, harmful dissolved oxygen concentrations have been reported in most other hurricane impact studies (Hagy et al. 2006, Tomasko et al. 2006, O'Connell et al. 2013). Salinity can increase or decrease depending on the track of the storm and associated wind and rainfall (Chen et al. 2008). Salinity increases from storm surges were found after most hurricanes we studied, but Tropical Storm Al-

lison decreased salinity in 2001. We used mean bottom salinity taken at the time of clam collection in this study because it was often higher than surface values. Increased salinity did not result in as much *R. cuneata* loss as it did during the La Niña drought, because after hurricanes, salinity increases and introduction of higher salinity species were temporary, compared to the multiple year effects of the prolonged drought, which produced a community shift that increased competition and resulted in *R. cuneata* decline. However, storm surges did increase bottom salinity which caused stratification in the water column and harmful hypoxic bottom waters. Poirrier et al. (2008) reported salinity stratification and low bottom dissolved oxygen after Hurricane Katrina. Poirrier et al. (2013) found salinity stratification and hypoxia at three sites south of Mandeville on 8 September 2012, 11 days after Hurricane Isaac's landfall.

Lack of complete recovery from the La Niña drought, Bonnet Carré Spillway openings, and saltwater intrusion from the MRGO also appears to have exacerbated hurricane effects. The spring 2008 Bonnet Carré Spillway opening, which filled Lake Pontchartrain with nutrient rich Mississippi River water, eliminated higher salinity species and introduced nutrients, which caused eutrophication in Mississippi and Chandeleur Sounds in the lower Basin. Storm surges from Ike and Gustav introduced higher salinity water that had become hypoxic from the Bonnet Carré Spillway opening, which affected the remaining lower salinity species. This synergistic effect likely produced stress which was greater than what would have been caused by either disturbance alone. Storm surges from Katrina, Rita, Ike and Gustav were severe because they also entered through the MRGO and the IHNC. Effects of Isaac were not as severe as these 2005 and 2008 storms because it was less intense and occurred after the closure of the MRGO (Shaffer et al. 2009, Poirrier 2013).

An obvious question is, have storm effects on *R. cuneata* clams recently increased? In 1998, Tropical Storms Frances and Hermine, and Hurricanes Earl and Georges affected southeastern Louisiana. No significant effects on clams from Georges were detected in 1998 and 1999 surveys. However, Hurricane Georges caused a 40% loss of land area in the Chandeleur Islands (Penland et al. 2001). We propose that coastal erosion caused by Hurricane Georges and other 1998 storms, adverse effects of the drought on wetlands (Visser et al. 2002), and increased sea level set the stage for increased effects of later storm surges on Lake Pontchartrain *R. cuneata* clams. From 1961–2000, the number and intensity of U.S. hurricanes decreased. After 2000, there was a period of increased hurricane frequency (Fearnley et al. 2009). Relative sea level rise in Louisiana is occurring 10 times faster than the global average and more than 5 times

faster than the Gulf of Mexico average (Penland and Ramsey 1990). Rates are high near the mouth of Lake Pontchartrain which is located in an abandoned, deteriorating Mississippi River Delta. Two sites in eastern Lake Pontchartrain have experienced relative sea level rise rates of 1 cm/yr (South Point 1949–1986; Little Woods 1931–1977; Penland and Ramsey 1990). This rate is comparable to that in Grand Isle, LA which has increased 0.93 cm/year (Burkett 2008). These factors should increase in the future as global climate changes.

At some point in time a tipping point for the loss of *R. cuneata* clams should occur due to increasing sea level (Burkett 2008), barrier island loss (Penland et al. 2001), wetland loss (463 km², 1932–2010; Couvillion et al. 2011), and possible enlargement of tidal passes (Li et al. 2009). We are currently, or will soon be, at this tipping point. We found clam recovery by 2014, but in the near future hurricanes may produce an irreversible decline due to direct surge damage, burial in sediment, anoxia and hypoxia from salinity stratification produced by the surge, and community changes from a long-term shift to higher salinity conditions.

CONCLUSIONS

Abrupt population changes in *R. cuneata* related to climate change occurred in Lake Pontchartrain, Louisiana. A 96% decrease occurred in 2001 after a drought from an ENSO shift (1998–2000). This decrease was caused by a salinity increase during the drought. More importantly, however, long periods without salinity decreases appeared to result in the establishment of higher salinity communities which, through competition, may have resulted in the *R. cuneata* decline. *Ischadium recurvum* colonized live *R. cuneata* and its overgrowth may have further influenced the *R. cuneata* decline. Rapid recovery following the drought did not occur due to a period of high hurricane frequency and intensity. After years of monitoring, we found density and biomass recovery in 2014. Recovery indicated that clam loss was not permanent, but caused by a series of climatic disturbances. Hurricane surges resuspended sediments which buried clams and abruptly increased salinity and lowered dissolved oxygen. Salinity stratification persisted after storms and caused hypoxia in bottom waters. Storm surge interactions with ENSO shifts, periodic operation of the Bonnet Carré Spillway, and the MRGO also intensified hurricane effects. Results of this study could be used to better understand the potential effects of global climate changes in the oligohaline zone of other Gulf of Mexico estuaries.

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LITERATURE CITED

- Abadie, S.W. and M.A. Poirrier. 2000. Increased density of large *Rangia* clams in Lake Pontchartrain after the cessation of shell dredging. *Journal of Shellfish Research* 19:481–485.
- Bedford, W.B. and J.W. Anderson. 1972. The physiological response of the estuarine clam *Rangia cuneata* (Grey) to salinity. I. Osmoregulation. *Physiological Zoology* 45:255–260.
- Boesch, D. 1977. A new look at the zonation of benthos along the estuarine gradient. In: B.C. Coull, ed. *Ecology of Marine Benthos*, University of South Carolina Press, Columbia, SC, USA, p. 245–266.
- Brammer, A.J., Z.R. del Rey, E.A. Spalding, and M.A. Poirrier. 2007. Effects of the 1997 Bonnet Carré Spillway opening on infaunal macroinvertebrates in Lake Pontchartrain, Louisiana. *Journal of Coastal Research* 23:1292–1303. DOI: 10.2112/05–0571.1
- Burkett, V. 2008. The northern Gulf of Mexico coast: Human development patterns, declining ecosystems, and escalating vulnerability to storms and sea level rise. In: M.C. MacCracken, F. Moore, and J.C. Topping Jr., eds. *Sudden and Disruptive Climate Change: Exploring the Real Risks and How we can Avoid Them*. Earthscan Publications, Sterling, VA, USA, p. 101–118.
- Cain, T.D. 1975. Reproduction and recruitment of brackish water clam *Rangia cuneata* in James River, Virginia. *Fishery Bulletin* 73:412–430.
- Carriker, M.R. 1967. Ecology of estuarine benthic invertebrates: A perspective. In: G.H. Lauff, ed. *Estuaries*. American Association for the Advancement of Science, Washington, D.C., USA, p. 442–487.
- Chen, Q., L. Wang, and R. Tawes. 2008. Hydrodynamic response of northeastern Gulf of Mexico to hurricanes. *Estuaries and Coasts* 31:1098–1116. DOI: 10.1007/s12237–008–9089–9.
- Cho, H.J. and M.A. Poirrier. 2005. Response of submersed aquatic vegetation (SAV) to the 1997–2001 El Niño Southern Oscillation shifts in Lake Pontchartrain, Louisiana. *Estuaries* 28:215–225. DOI: 10.1007/BF02732856
- Coleman, E., 2003. The Gulf Oyster Industry: Seizing a better future. Gulf Oyster Industry Program. National Seagrass College Program, Baton Rouge, LA, USA, p. 1–14.
- Couvillion, B.R., J.A. Barras, G.D. Steyer, W. Sleavin, M. Fischer, H. Beck, N. Trahan, B. Griffin, and D. Heckman. 2011. Land area change in coastal Louisiana from 1932 to 2010: U.S. Geological Survey Scientific Investigations Map 3164, scale 1:265,000, Reston, VA, USA, 12 p. pamphlet.
- Edmondson, W.T. 1971. Methods for processing samples and developing data. In: W.T. Edmondson and G.G. Winberg, eds. *A Manual on Methods for the Assessment of Secondary Productivity in Fresh Waters*. International Biological Programme Handbook No. 17. Blackwell Scientific Publications, Oxford, England, p. 127–169.
- Engle, V.D., J.L. Hyland, and C. Cooksey. 2009. Effects of Hurricane Katrina on benthic macroinvertebrate communities along the northern Gulf of Mexico coast. *Environmental Monitoring and Assessment* 150:193–209. DOI: 10.1007/s10661–008–0677–8
- Fairbanks, L. 1963. Biondemographic studies of the clam *Rangia cuneata* Gray. *Tulane Studies in Zoology* 10:3–47.
- Fearnley, S.M., M.D. Miner, M. Kulp, C. Bohling, and P.S. Penland. 2009. Hurricane impact and recovery shoreline change analysis of the Chandeleur Islands, Louisiana, USA: 1855–2005. *Geo-Marine Letters* 29:455–466. DOI: 10.1007/s00367–009–0155–5
- Flocks, J., J. Kindinger, M. Marot, and C. Charles. 2009. Sediment characterization and dynamics in Lake Pontchartrain, Louisiana. *Journal of Coastal Research* SI 54:113–126. DOI: 10.2112–SI54–011.1
- Flowers, G.C. and W.C. Isphording. 1990. Environmental sedimentology of the Pontchartrain Estuary. *Transactions Gulf Coast Association Geological Society* 40:237–250.
- Francis, J.C., M.A. Poirrier, D.E. Barbe, V. Wijesundera, and M.M. Mulino. 1994. Historic trends in secchi disk transparency of Lake Pontchartrain. *Gulf Research Reports* 9:1–16. DOI: 10.18785/grr.0901.01
- Gotelli, N.J. and A.M. Ellison. 2004. *A Primer of Ecological Statistics*. Sinauer Associates, Inc., Sunderland, MA, USA, 510 p.
- Hagy, J.D., J.C. Lehrter, and M.C. Murrell. 2006. Effects of Hurricane Ivan on water quality in Pensacola Bay, Florida. *Estuaries and Coasts* 29:919–925. DOI: 10.1007/BF02798651
- Houck, O.A., F. Wagner, and J.B. Elstrott. 1989. To restore Lake Pontchartrain: A report to the Greater New Orleans Expressway Commission on the sources, remedies, and economic impacts of pollution in the Lake Pontchartrain Basin. Metairie, LA, USA. 270 p.
- LaSalle, M.W. and A.A. de la Cruz. 1985. Species profiles: life histories and environmental requirements of coastal fisheries and invertebrates (Gulf of Mexico) – common *Rangia*. U.S. Fish and Wildlife Service Biological Report 82(11.31). U.S.

- Army Corps of Engineers, Vicksburg, MS, USA, TR EL-82-4, 16 p.
- Li, C., E. Weeks, and J.L. Rego. 2009. In situ measurements of saltwater flux through tidal passes of Lake Pontchartrain estuary by Hurricanes Gustav and Ike in September 2008. *Geophysical Research Letters* 36:L19609. DOI: 10.1029/2009GL039802
- Neter, J., W. Wasserman, and M.H. Kutner. 1990. *Applied linear statistical models*. Irwin/Mc-Graw-Hill, Homewood, IL, USA, 1184 p.
- O'Connell, M.T., A.M. O'Connell, and C.S. Schieble. 2013. Response of Lake Pontchartrain fish assemblages to Hurricanes Katrina and Rita. *Estuaries and Coasts* 37:461-475. DOI: 10.1007/s12237-013-9675-3
- Penland, S. and K.E. Ramsey. 1990. Relative sea-level rise in Louisiana and the Gulf of Mexico: 1908-1988. *Journal of Coastal Research* 6:323-342.
- Penland, P.S., P. McCarty, A. Beall, and D. Maygarden. 2001. Regional description of the Lake Pontchartrain Basin. In: P.S. Penland, A. Beall, and J. Waters, eds. *Environmental Atlas of the Lake Pontchartrain Basin*, Lake Pontchartrain Basin Foundation, New Orleans, LA, USA, p. 3-6.
- Poirrier, M.A. 1978. Studies of salinity stratification in southern Lake Pontchartrain near the Inner Harbor Navigational Canal. *The Proceedings of Louisiana Academy of Sciences* 41:26-35.
- Poirrier, M.A. 2013. Effects of closure of the Mississippi River Gulf Outlet on saltwater intrusion and bottom water hypoxia in Lake Pontchartrain. *Gulf and Caribbean Research* 25:105-109. DOI: 10.18785/gcr.2501.07
- Poirrier, M.A., T.M. Soniat, Y. King, and L. Smith. 1984. An evaluation of the southern Lake Pontchartrain benthos community. 64003-84-05. Final Report. Louisiana Department of Environmental Quality Office of Water Resources Water Pollution Control Division, New Orleans, LA, USA. p 1-79.
- Poirrier, M.A., Z.R. del Rey, and E.A. Spalding. 2008. Acute disturbance of Lake Pontchartrain benthic communities by Hurricane Katrina. *Estuaries and Coasts* 31:1221-1228. DOI: 10.1007/s12237-008-9103-2
- Poirrier, M.A., E.A. Spalding, and C.D. Franze. 2009. Lessons learned from a decade of assessment and restoration studies of benthic invertebrates and submersed aquatic vegetation in Lake Pontchartrain. *Journal of Coastal Research* SI 54:88-100. DOI: 10.2112/SI54-005.1
- Poirrier, M.A., C.N. Dunn, C.E. Caputo, G.C. Flowers, and J.M. Adams. 2013. The role of ENSO Climate Shifts and the increase in the frequency and intensity of storm surges in the decline of large *Rangia cuneata* clams in Lake Pontchartrain. *Basics of the Basin* 2013:177-81. www.saveourlake.org/PDF-documents/BOB/BasicsoftheBasinProceedings2013.pdf. (viewed on 02/04/2015).
- Ray, G.L. 2009. Response of benthic invertebrate communities following the 2008 Bonnet Carré Spillway release. A Report to the U.S. Army Engineer District, New Orleans. U.S. Army Engineer Research and Development Center, Vicksburg, MS, USA, 150 p.
- Remane, A. and C. Schlieper. 1971. *Biology of Brackish Water*. Wiley Interscience Division: John Wiley & Sons, Inc., New York, NY, USA. 372 p.
- Roth, D. 2010. *Louisiana Hurricane History*. National Weather Service, Camp Springs, MD, USA, 70 p.
- Saucier, R.T. 1963. Recent geomorphic history of the Pontchartrain Basin. Louisiana State University Press, Baton Rouge, LA, USA, 114 p.
- Shaffer, G.P., J.W. Day, S. Mack, G.P. Kemp, I. van Heerden, M.A. Poirrier, K.A. Westphal, D. FitzGerald, A. Melanes, C.A. Morris, R. Bea, and P.S. Penland. 2009. The MRGO Navigation Project: A massive human-induced environmental, economic, and storm disaster. *Journal of Coastal Research* SI 54:206-224. DOI: 10.2112/SI54-004.1
- Sikora, W.B. and B. Kjerfve. 1985. Factors influencing the salinity regime of Lake Pontchartrain, Louisiana, a shallow coastal lagoon: analysis of a long-term data set. *Estuaries* 8:170-180. DOI: 10.2307/1351866
- Sikora, W.B. and J.P. Sikora. 1982. Ecological characterization of the benthic community of Lake Pontchartrain, Louisiana. LSU-CEL-82-05. Final Report. U.S. Army Corps of Engineers, New Orleans District, New Orleans, LA, USA. 214 p.
- Spalding, E.A., A.E. Walker, and M.A. Poirrier. 2007. Restoration of 100 square miles of shellfish habitat in Lake Pontchartrain. GMP: MX974852-03-0. Final Report. Gulf of Mexico Program Environmental Protection Agency, Stennis Space Center, MS, USA. 26 p.
- Suttkus, R.D., R.M. Darnell, and J.H. Darnell. 1954. Biological study of Lake Pontchartrain. In: Tulane University Annual Report 1953-54. New Orleans, LA, USA, 59 p.
- Swenson, E.M. 1980. General hydrography of Lake Pontchartrain, Louisiana. In: J.H. Stone, ed. *Environmental analysis of Lake Pontchartrain, Louisiana, its surrounding wetlands, and selected land uses*. Vols. 1 and 2. LSU-CEL-80-08. Final Report. Center for Wetland Resources, Louisiana State University, Baton Rouge, LA, USA. p. 57-156.
- Tarver, J.W. and R.J. Dugas. 1973. A study of the clam, *Rangia cuneata*, in Lake Pontchartrain and Lake Maurepas, Louisiana. Project 2-91-R. Technical Bulletin. Louisiana Wild Life and Fisheries Commission, New Orleans, LA, USA. 97 p.
- Tomasko, D.A., C. Anastasiou, and C. Kovach. 2006. Dissolved oxygen dynamics in Charlotte Harbor and its contributing watershed, in response to Hurricanes Charley, Frances, and Jeanne - impacts and recovery, 1985-2004. *Estuaries and Coasts* 29:932-938. DOI: 10.1007/BF02798653
- U.S. Army Corps of Engineers. 1987. Clam shell dredging in Lakes Pontchartrain and Maurepas, Louisiana. ADA186642. Final Report, U.S. Army Corps of Engineers, New Orleans, LA, USA. 151 p.
- Verween, A., F. Kerckhof, M. Vincs, and S. Degraer. 2006. First European record of the invasive brackish water clam *Rangia*

- cuneata* (G. B. Sowerby I, 1831) (Mollusca: Bivalvia). Aquatic Invasions 1:198–203. DOI: 10.3391/ai.2006.1.4.1
- Visser, J.M., C.E. Sasser, R.H. Chabreck, and R.G. Linscombe. 2002. The impact of a severe drought on the vegetation of a subtropical estuary. Estuaries 25:1184–1195. DOI: 10.1007/BF02692215
- Zheng, D., X. Ding, Y. Zhou, and Y. Chen. 2003. Earth rotation and ENSO events: combined excitation of interannual LOD variations by multiscale atmospheric oscillations. Global and Planetary Change 36:89–97. DOI: 10.1016/s0921–8181(02)00176–5
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SURVEY OF TWO PERFLUORINATED ORGANIC COMPOUNDS (PFOA AND PFHxA) IN WATER AND BIOTA SURROUNDING A POLYFLUORINATED CHEMICAL PLANT

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ABSTRACT: Perfluorooctanoic acid (PFOA) and perfluorohexanoic acid (PFHxA) concentrations were measured in water and biota samples collected within and near a polyfluorinated chemical plant in coastal Mississippi. Effluents from the site and from the local public water treatment plant were sampled along with municipal water supplies, shallow groundwater beneath the site, nearby surface waters, and local biota. Highest concentrations were from stormwater ([PFOA] = 85–530 ng/l; [PFHxA] = 140–590 ng/l) and shallow groundwaters ([PFOA] = 44–1000 ng/l; [PFHxA] = 210–3100 ng/l) collected at the site. The local public water treatment effluent also had relatively high PFHxA concentrations (310–590 ng/l). Intermediate PFOA concentrations were measured in effluent samples (21–33 ng/l) and irrigation water from a public works facility (15–48 ng/l). Drinking water and all surface waters had low PFOA and PFHxA concentrations (PFOA = 1.1–3.0 ng/l; PFHxA < 1.4 ng/l). Tissues from local fish and blue crab had undetectable PFOA and PFHxA concentrations. Results suggest that evaporation and subsequent percolation of contaminated stormwater into the shallow aquifer is the major pathway for perfluorinated contaminants to escape from this site.

KEY WORDS: Perfluorooctanoic acid, C8, perfluorohexanoic acid, stormwater, groundwater

INTRODUCTION

Perfluorinated organic compounds (PFCs) were first manufactured in the 1940's and have been used in a variety of applications ranging from providing water and oil repellency for a multitude of surfaces (clothing, furniture, floor waxes, etc) to its use in aircraft production, electronics, personal care products, cookware and food packaging products. These compounds are of particular concern due to their ubiquity and persistence in the environment and their bioaccumulation in wildlife and humans. The average PFOA concentration in American adults' blood serum is 4 ng/ml (Calafat et al. 2007) while much higher concentrations (> 76 ng/ml) have been observed in other populations exposed to contaminated drinking waters (Bartell et al. 2010; Hoffman et al. 2011). PFCs are suspected to be carcinogenic and may also act as endocrine disruptors (White et al. 2011). In an effort to better understand the environmental and organismal impacts of such compounds, many studies have assessed contamination in water supplies (Post et al. 2009), wastewater treatment plants (Sinclair and Kannan 2006; Guo 2010), and a wide variety of environmental samples including fresh and marine waters (Kallenborn et al. 2004, Yamashita et al. 2005, Tanaka et al. 2006, Skutlarek et al. 2006, deVoogt et al. 2006, Weremiuk et al. 2006, Loos et al. 2007), stormwater (Murakami et al. 2009a, Xiao et al. 2011, Nguyen et al. 2011), groundwaters (Shultz et al. 2004, Murakami et al. 2009b) and biota ranging from small invertebrates (vanLeeuwen et al. 2006, Gulkowska et al. 2006, Nakata et al. 2006) to large mammals in a wide variety of habitats (Giesy and Kannan 2001). Few studies have examined a wide variety of sample types (municipal waters, natural waters and biota) collected within and near a single manufacturing plant.

The First Chemical Corporation (FCC) in Pascagoula,

MS uses several industrial processes to purify fluorotelomer alcohols as precursors to the manufacture of surfactants and ingredients for fire-fighting foam. This study was conducted on samples collected during 2008 and 2010, as a follow up to the 2006 Pascagoula baseline study conducted by FCC to provide a snapshot of perfluorooctanoic acid (PFOA) concentrations in local waters and biota prior to the startup of a new fluorotelomer alcohol purification operation at the site. The manufacturing of 8–2 fluorotelomer alcohol, a precursor of PFOA, is currently being phased out and being replaced by a process that exclusively produces 6–2 fluorotelomer alcohol (6–2 FTOH). Therefore, analysis of perfluorohexanoic acid (PFHxA), which is both a potential low level impurity and degradation product of 6–2 FTOH, was also conducted for samples in this study. Industrial effluents from the chemical plant are discharged to the Pascagoula Publicly-Operated Treatment Works (POTW) wastewater treatment facility which in turn discharges its treated effluent to the Pascagoula River. The objective of this study was to report concentrations of 2 PFCs within and around an operational perfluorinated chemical plant and in local municipal waters (i.e., water supply and wastewater treatment plant effluent) and biota.

MATERIALS AND METHODS

Sample Collections

To examine sources of PFOA and PFHxA, I collected samples from a variety of locations in 2008 and 2010, including the effluent from the FCC to the POTW, the POTW effluent discharged to the Pascagoula River, and solid wastewater sludge from the Pascagoula POTW. Also, stormwater and groundwater samples were collected on the chemical

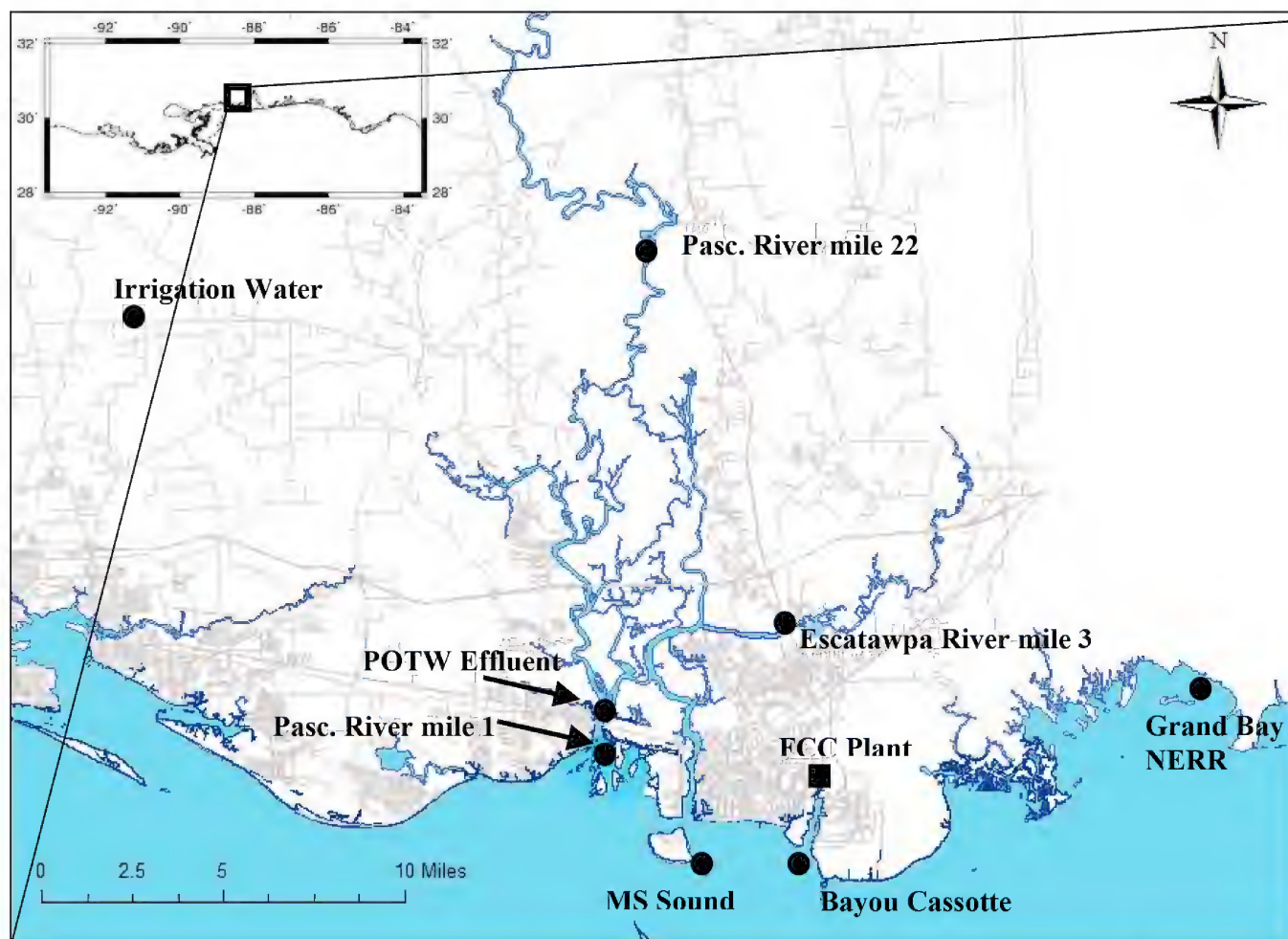


FIGURE 1. Map of the study area near Pascagoula, Mississippi showing the location of the chemical plant (square symbol ■) and all sampling locations (black circles ●). Biota were sampled at the 3 estuarine sites.

plant site and municipal water samples were collected to evaluate background concentrations of the local drinking water supply and irrigation water used at the West Jackson County Landfarm which originates from the nearby City of Ocean Springs POTW. Consistent with the FCC 2006 baseline study, a variety of surface water samples were collected from locations in nearby rivers and in Mississippi Sound (Figure 1). The Pascagoula River was sampled 34 km upstream and 2 km downstream of the Pascagoula POTW discharge (Figure 1). An additional river sample was collected from the Escatawpa River before its confluence with the Pascagoula River. Estuarine samples included a site outside of Bayou Cassotte, a heavily industrialized bayou near the FCC plant; a site in Grand Bay, a nearby NOAA National Estuarine Research Reserve; and a site in Mississippi Sound near the mouth of the Pascagoula River. Hardhead Catfish (*Ariopsis felis*), Spotted Seatrout (*Cynoscion nebulosus*) and the blue crab (*Callinectes sapidus*) were collected in 2008 at the same locations as the 2006 Pascagoula baseline study. Biota sampling sites corresponded to water sampling sites (Figure 1). All samples were collected from mid-July to

early September, consistent with the 2006 baseline study methodology.

Effluent samples

FCC effluent samples were provided by FCC, as a QA/QC split from their weekly monitoring. This sample was composited from hourly sub-samples taken over a 24-hour period. Effluent samples from the Pascagoula POTW were collected on 18 August 2008 and 21 August 2010. Irrigation water samples were taken from a spigot at the West Jackson County Landfarm on 31 July 2008 and 1 September 2010. This effluent comes from the City of Ocean Springs POTW and is not connected to the Pascagoula water supply or the Pascagoula POTW.

POTW Sludge Sampling

Sludge samples were collected from the Pascagoula POTW on 18 August 2008 and 21 August 2010 from the top 5 cm of the sludge bed, placed in a clean aluminum mixing bowl, and homogenized in the field. In 2010 duplicate field samples were collected within 1 m of each other. Sample bottles were filled, placed on ice and shipped to the contract laboratory.

Water Samples

Storm-water samples were taken from a stormwater trough at the Pascagoula plant on 11 August 2008 and 12 August 2010 after rain events by submerging the sample bottle and capping while submerged to avoid sampling surface microlayer particles. Groundwater was sampled from three pre-existing shallow groundwater wells (screened from 4.3 to 4.9 m) at the FCC site on 30 July 2008 and 12 August 2010 near a new chemical process unit (Figure 2). Prior to sampling, each monitor well was purged of 3 well volumes using a peristaltic pump with aged and pre-cleaned Tygon tubing. None of the wells were pumped dry during the purging and sampling. Water from this unconfined sand aquifer is not a potable water source and is isolated from deeper aquifers by a clay confining layer. Samples were collected at monitoring wells MW-17 and MW-63 near the new chemical process unit whereas monitor well MW-28 was down gradient from the process unit (Figure 2).

Municipal drinking water samples from the Jackson County Port Authority were collected directly from taps at the chemical plant on 31 July 2008 and on 12 August 2010. Surface water samples were collected from the Pascagoula and Escatawpa Rivers, Bayou Cassotte, Grand Bay and the Mississippi Sound (Figure 1) during flood and ebb tides at approximately mid-stream and mid-depth using an aged,

pre-cleaned acrylic horizontal Beta water sampler (Wildco part number 1940-G64). All water samples were collected in pre-cleaned and pre-labeled bottles and stored on ice and cold-shipped to the contract laboratory overnight.

Biota samples

Hardhead Catfish and Spotted Seatrout were collected at the Grand Bay and Mississippi Sound stations with gill nets deployed for 2 hours on 8, 15, 20, 22, and 28 August 2008. Six fish of each species were collected, ranging in size from 23–43 cm total length (TL). The captured fish were placed in a live well, whereas by-catch species or fish not meeting the minimum size requirements were returned to the water to avoid injury. The selected fish were euthanized with MS222 and placed in Ziploc™ bags on ice until tissues were processed. Fillets and carcass (viscera, liver, and skin) were separated, sealed in Ziploc™ bags, and frozen at -20°C . Blue crabs were collected from the same sites with a baited crab trap deployed over a tidal cycle. Six legal-sized crabs (> 12.7 cm carapace width) were placed directly on ice, whereas by-catch and crabs not meeting the minimum size requirements were returned to the water as soon as possible. Soft tissues samples were separated from the crabs and placed in Ziploc™ bags and frozen (-20°C). Tissue samples were shipped on dry ice overnight to the contract laboratory.

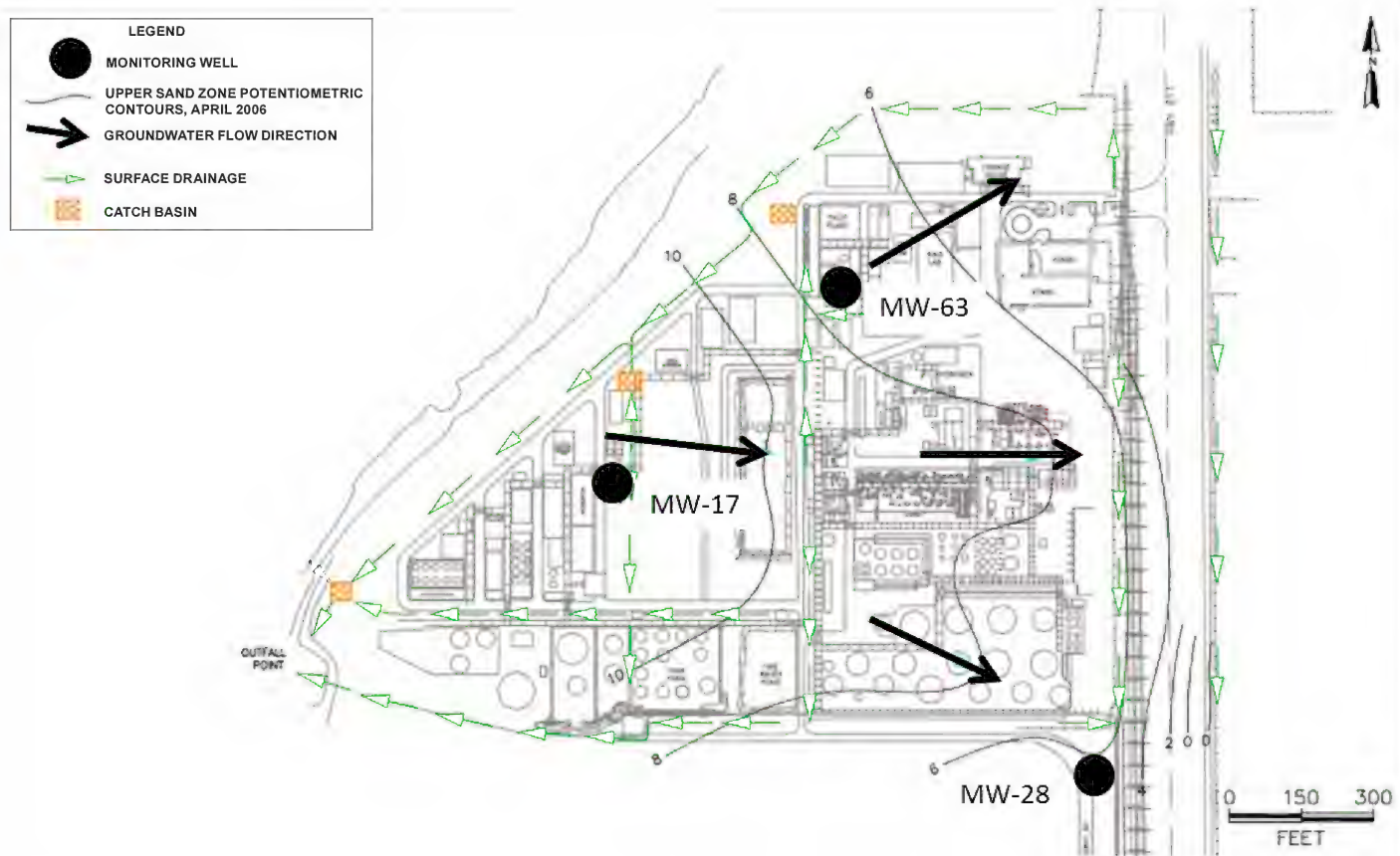


FIGURE 2. Site map of the chemical plant showing the location of the 3 monitoring wells used to collect groundwater samples. Arrows denote the primary direction of groundwater movement based on the hydraulic gradient of the water table across the site during the Pascagoula Baseline Study

Analytical Methods

Water samples from 2008 and 2010 were analyzed by Axys Analytical Services, Ltd (Sidney, British Columbia, Canada) for PFOA and PFHxA concentrations. Since a different lab, MPI Research (formerly Exygen; State College, Pennsylvania), analyzed water samples from FCC's 2006 baseline study, duplicate water samples were collected in 2008 and sent to MPI Research to compare results from the two labs. Sample preparation, instrumental analysis and analyte quantification procedures followed Axys Method MU-060 *Analytical Procedure for the Analysis of Perfluorinated Organics Compounds in Aqueous Samples* by LC-MS/MS. Accurately weighed samples (about 500 g) were spiked with ^{13}C -labelled quantification standards, extracted and cleaned using solid phase extraction (SPE) cartridges. Target analytes were extracted in basic methanol. After being spiked with labeled recovery (internal) standards, the extract was analyzed using HPLC-MS/MS. Analyte concentrations were determined by isotope dilution/internal standard method, comparing the area of the quantification ion to that of the ^{13}C -labelled quantification standard and correcting for response factors. The limit of detection (LOD) and limit of quantification (LOQ) for both PFOA and PFHxA ranged from 0.99–1.9 for PFOA and 1.0–1.4 ng/l for PFHxA during 2008. The LOD and LOQ for PFOA and PFHxA during 2010 ranged from 1.0–2.6 ng/l.

For sludge and tissue samples, sample preparation and analyte quantification procedures followed Axys Method MLA-041: *Analytical Procedure for the Analysis of Perfluorinated Organic Compounds in Solid Samples* by LC-MS/MS. Accurately weighed samples (about 5 g dry) were spiked with ^{13}C -labelled quantification standards and extracted in acetic acid and basic methanol. The extract was purified using SPE cartridges and activated carbon. The extract was analyzed using HPLC-MS/MS with the isotope dilution/internal standard method by comparing the area of the quantification ion to that of the ^{13}C -labelled quantification standard and correcting for response factors. The LOD for this method was 1.9–2.5 ng/g.

RESULTS AND DISCUSSION

Effluents

The PFOA concentrations of effluent from the chemical plant were 38 ng/l in 2008 and 13 ng/l in 2010, similar to that measured during the 2006 baseline study (Table 1). The PFHxA concentrations were 590 ng/l in 2008 and 310 ng/l in 2010. The PFOA concentrations in the POTW effluent from 2006 to 2010 were similar to the FCC effluent (21 ng/l–33 ng/l). The PFOA concentrations of the Ocean Springs POTW effluent water used for irrigation at the Jackson County Landfarm were 11 ng/l in 2006, 48 ng/l in 2008 and 16 ng/l in 2010. The PFHxA concentrations in this irrigation water were 22 ng/l and 8 ng/l during 2008

and 2010, respectively. This irrigation water is not connected to the Pascagoula water supply or the Pascagoula POTW. These results appear to be within the normal variability of PFOA concentrations in these different effluents, which are all lower than PFOA concentrations measured in 6 U.S. city wastewater effluents (40 ng/l–2420 ng/l; 3M Environmental Laboratory 2001). The PFHxA concentrations were on the low end of measurements from 19 different wastewater treatment plants in Korea (3.4 ng/l–591 ng/l; Guo et al. 2010). Sewage treatment plants can be a source of PFOA if the influent stream contains significant concentrations of PFOA precursors (Sinclair and Kannan 2006). Guo et al. (2010) found that the concentrations of PFCs in POTW were highest in large cities of Korea, suggesting that household sewage is likely a larger source of some PFC's to the environment than industrial wastewaters, which had high concentrations but small release volumes.

Sludge samples

The PFOA concentrations in sludge samples from 2008 and 2010 were similar (32 $\mu\text{g/kg}$ and 33 $\mu\text{g/kg}$, respectively) and were lower than during the 2006 baseline study (68 $\mu\text{g/kg}$; Table 1). The PFHxA concentrations in the sludge were 6.6 $\mu\text{g/kg}$ in 2008 and 2.5 $\mu\text{g/kg}$ in 2010. The PFOA concentrations from the wastewater sludge are on the low end of the range reported in sludge from 6 U.S. cities which ranged from < 0.2 $\mu\text{g/kg}$ –244 $\mu\text{g/kg}$ (3M Environmental Laboratory 2001). Guo et al. (2010) found PFOA concentrations in wastewater sludge ranged from below detection (<4.0 $\mu\text{g/kg}$) to 24.7 $\mu\text{g/kg}$ in Korea while PFHxA concentrations were all below detection (<4 $\mu\text{g/kg}$).

Water Samples

The PFOA concentrations measured in water samples by MPI and Axys labs were very highly correlated ($y = 1.01x + 8.7$; $r^2 = 0.996$, where x and y are the PFOA concentrations from MPI and Axys, respectively), showing good agreement among these independent laboratory analyses (Table 1). The PFOA concentrations of stormwater samples collected in 2008 and 2010 were 530 ng/l and 85 ng/l, respectively, while PFHxA concentrations in stormwater in 2008 and 2010 were 590 ng/l and 140 ng/l, respectively. These stormwater concentrations were much higher than the effluent from the chemical plant, suggesting a source of PFOA within the plant grounds, such as from a leak or from released aerosols that settled onto surfaces and entrained into stormwater effluent. These measured concentrations are high relative to other studies that examined stormwater, although most other studies sampled in urbanized areas rather than from a chemical plant site. For example, stormwater from Minnesota's Twin Cities metropolitan area had a mean PFOA concentration of 9.3 ng/l (Xiao et al. 2011) while an urbanized region of Singapore had PFOA concentrations of 5 ng/l – 31 ng/l (Nguyen et al 2011), and stormwater runoff in Tokyo had PFOA concentrations ranging

TABLE 1. Perfluorooctanoic acid (PFOA) concentrations in 2006, 2008, and 2010 and perfluorohexanoic acid (PFHxA) concentrations in 2008 and 2010 for water samples collected near a chemical plant in Pascagoula, MS during this study. Number in parenthesis shows minimum limit of detection (ND) or quantification (NQ).

Sample location	2006* PFOA (ng/l)	2008a** PFOA (ng/l)	2008b PFOA (ng/l)	2010 PFOA (ng/l)	2008 PFHxA (ng/l)	2010 PFHxA (ng/l)
Effluent POTW	33	17	21	33	22	14
Effluent POTW lab dup		16				
Effluent FCC	10	40	38	13	590	310
Effluent FCC lab dup		45				
Effluent FCC - field dup	11	49				
Stormwater FCC	460	480	530	85	590	140
Stormwater FCC lab dup		500				
Stormwater FCC - field dup		700				
Irrigation water - Ocean Springs POTW	11	43	48	15	22	7.3
Irrigation water - Ocean Springs POTW - field dup				17		8.6
Pascagoula drinking water	ND (1.6)	ND	2.8	ND	1.6	ND
Pascagoula drinking water - field dup	ND (1.6)	ND	2.5	1.4	1.7	ND
Monitor well 17	44	1000	1000	280	790	520
Monitor well 28	NQ (7.8)	94	82	250	210	360
Monitor well 28 lab dup		91				
Monitor well 63	79	250	290	940	810	2900
Monitor well 63 - field dup		260		950		3100
Pascagoula River upstream high tide	NQ (7.8)	ND	2.9	1.7	3.7	ND
Pascagoula River upstream high tide - lab dup				1.5		ND
Pascagoula River upstream low tide	NQ (7.8)	ND	2.2	1.5	2.1	ND
Pascagoula River downstream high tide	NQ (13)	ND	1.8	2.1	ND	ND
Pascagoula River downstream low tide	NQ (13)	ND	2.4	2.2	1.3	ND
Escatawpa River high tide	NQ (13)	ND	1.6	1.7	2	ND
Escatawpa River low tide	NQ (13)	ND	2.6	1.7	1.5	ND
Bayou Cassotte high tide	NQ (13)	ND	1.2	2.2	ND	1.4
Bayou Cassotte low tide	NQ (13)	ND	1.7	2.1	1.5	1.6
Mississippi Sound high tide	NQ (13)	ND	2.1	1.7	1.6	ND
Mississippi Sound low tide	NQ (13)	ND	1.8	2.2	ND	ND
Grand Bay high tide	NQ (7.8)	ND	2.2	2.9	1.7	1.3
Grand Bay high tide lab dup				2.5		1.4
Grand Bay low tide	ND (1.6)	ND	3	2.7	2.8	ND
	(µg/kg)		(µg/kg)		(µg/kg)	
Sludge - Pascagoula POTW	68		32	33	6.6	2.5
Sludge - Pascagoula POTW - field dup	67			34		2.5

*Data from 2006 was provided by the First Chemical Corporation's Pascagoula Baseline study.

** Results from MPI Research Labs

from 15 ng/l to about 105 ng/l (Murakami et al. 2009a).

Groundwater collected during 2008 from monitor well MW-17 (Figure 2) contained the highest PFOA concentration (1000 ng/l) of all water samples in this study (Table 1). Groundwater from MW-63 had a lower PFOA concentration (290 ng/l) and groundwater collected down gradient of the new chemical processing unit at MW-28 had still lower PFOA concentrations among lab and field duplicates (82–94 ng/l). During the 2010 sampling, the PFOA concentration in MW-17 was 280 ng/l while MW-28 and MW-63 had PFOA concentrations of 250 and 945 (± 5 ng/l), respectively. The PFHxA concentrations in MW-17

were 790 ng/l and 520 ng/l during 2008 and 2010, respectively. The PFHxA concentrations at MW-28 and MW-63 followed the same trend as the PFOA concentrations with lower concentrations during 2008 (210 and 810 ng/l, respectively) than during 2010 (360 ng/l and 3000 \pm 100 ng/l, respectively). Notably, the latter measurement was the highest PFHxA concentration observed for any water sample during the study. Published values of PFOA in shallow groundwater are sparse in the literature as most studies have justifiably focused on drinking water wells which often pump from deeper aquifers. However, groundwater from 33 m depth beneath metropolitan Tokyo had lower

PFOA concentrations (0.1 ng/l–94 ng/l; Murakami et al. 2009b) than those reported here. More extreme cases of PFC contamination have been observed in other shallow groundwaters where sources of PFCs are high. For example, extremely high PFOA concentrations were measured (up to 6.57×10^6 ng/l) in shallow groundwaters beneath US Air Force's fire training pads, where PFCs were in heavy use over long periods (Shultz et al. 2004).

The PFOA and PFHxA concentrations in drinking water supplied by the Jackson County Port Authority were some of the lowest observed in the study (<2.8 ng/l; Table 1). For comparison, PFOA concentrations in drinking water in 6 southern U.S. cities (Decatur, AL; Mobile, AL; Columbus, GA; Pensacola, FL; Cleveland, TN and Port St. Lucie, FL), was measured during 1999–2000 and found that it was only detectable (> 7.5 ng/l) in Columbus, GA where concentrations up to 27 ng/l were found (LOQ of 25 ng/l; 3M Environmental Laboratory 2001). Other studies have shown drinking water PFOA concentration in New Jersey (< 39 ng/l (Post et al. 2009) were similar to those typically measured in several Asian countries (< 0.1 – 41 ng/l; Saito et al. 2004; Tanaka et al. 2006), although sporadic concentrations from some sites were as high as 456 ng/l (So et al. 2007).

All 2008 and 2010 surface water samples from the Pascagoula River, Escatawpa River, Bayou Cassotte, Grand Bay, and the Mississippi Sound had PFOA concentrations that ranged from below detection (< 1.0 ng/l) to 3.0 ng/l (Table 1). The PFHxA concentrations were also undetectable or low in all surface water samples (< 1.0 – 3.7 ng/l). The surface water concentrations reported here are on the low end of those reported in other coastal and fresh waters (Kallenborn et al. 2004, Yamashita et al. 2005, Tanaka et al. 2006, Skutlarek et al. 2006, deVoogt et al. 2006, Weremiuk et al. 2006, Loos et al. 2007).

Biota

PFOA concentrations in all Hardhead Catfish, Spotted Seatrout, Eastern oyster (*Crassostera virginica*) and blue crab samples (fillets and viscera) were low in the 2006 Pascagoula baseline study (1.4–2.2 ng/g). Hardhead Catfish, Spotted Seatrout and blue crab all had undetectable concentrations of PFOA and PFHxA (< 2.5 ng/g) with the analytical limits of the analytical technique used in 2008. Oysters were not sampled during 2008. The 2006 baseline study results are on the low end of reported PFOA concentrations in fish

from other North American studies (1.0–44 ng/g; Martin et al. 2004, Furdui et al. 2007). Many studies in Europe and Asia have also reported a large percentage of undetectable concentrations in fish (57–85%; van Leeuwen et al. 2006, Gulkowska et al. 2006, Tseng et al. 2006) and crustaceans (30–100%; vanLeeuwen et al. 2006, Gulkowska et al. 2006, Nakata et al. 2006).

CONCLUSION

Cumulatively, the results of this study show that stormwater and shallow groundwater near the chemical plant appear to have elevated concentrations of PFOA and PFHxA. The PFCs from the FCC facility are likely entrained into stormwater where evaporation may concentrate PFCs, which then percolate into the shallow sand aquifer, thereby resulting in the high groundwater concentrations observed in this study. Short chain PFCs ($C \leq 8$), such as PFOA and PFHxA, do not appreciably bind to sediments relative to longer chain PFCs ($C \geq 11$), and thus are usually present in their dissolved form in field and laboratory settings (Higgins and Luthy 2006, Ahrens et al. 2010, Ahrens et al. 2011). PFOA has been shown to weakly sorb to sediments while PFHxA does not bind significantly (Chen et al. 2015). Thus, percolation into ground water appears to be the major pathway for perfluorinated chemicals to escape the production site. Groundwater flow rates (v) can be estimated across the site if the hydraulic conductivity (K) and the hydraulic gradient (dh/dl) are known using Darcy's Law:

$$v = K (dh/dl).$$

Calculated geometric means for K in the area ranged from 28–51 m/d (Prudic 1991) and the hydraulic gradient across the site during the 2006 baseline study was very small ($0.25 \text{ m} / 295 \text{ m} = 8.5 \times 10^{-4}$), resulting in a groundwater velocity of 2.4–4.3 cm/d (0.9–1.6 m/yr). While groundwater seepage and tidal exchange between groundwater and surface waters are likely to occur, the slow groundwater transport rate along with the microtidal nature of the Mississippi Sound results in a slow exchange rate of the contaminant plume with Mississippi Sound surface waters. Thus, the dilution of the contaminated groundwater seeping into Mississippi Sound by surface waters likely explains the low PFOA and PFHxA concentrations in nearby estuarine waters and biota.

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LITERATURE CITED

- 3M Environmental Laboratory. 2001. Executive Summary: Environmental Monitoring—Multi—City Study. Water, Sludge, Sediment, POTW Effluent and Landfill Leachate Samples, Minnesota 226–1030a. U.S. Environmental Protection Agency Docket, AR, USA, 12 p.
- Ahrens L., S. Taniyasu, L.W.Y. Yeung, N. Yamashita, P.K.S. Lam, and R. Ebinghaus. 2010 Distribution of polyfluoroalkyl compounds in water, suspended particulate matter and sediment from Tokyo Bay, Japan. *Chemosphere* 79:266–272. DOI:10.1016/j.chemosphere.2010.01.045
- Ahrens L., L.W.Y. Yeung, S. Taniyasu, P.K.S. Lam, and N. Yamashita. 2011. Partitioning of perfluorooctanoate (PFOA), perfluorooctane sulfonate (PFOS) and perfluorooctane sulfonamide (PFOSA) between water and sediment. *Chemosphere* 85:731–737. DOI:10.1016/j.chemosphere.2011.06.046
- Bartell S.M., A.M. Calafat, C. Lyu, K. Kato, P.B. Ryan, and K. Steenland. 2010. Rate of decline in serum PFOA concentrations after granular activated carbon filtration at two public water systems in Ohio and West Virginia, *Environmental Health Perspectives* 118:222–228. DOI:10.1289/ehp.0901252
- Calafat A.M., L.Y. Wong, Z. Kuklenyik, J.A. Reidy, and L.L. Needham. 2007. Polyfluoroalkyl chemicals in the U.S. population: data from the National Health and Nutrition Examination Survey (NHANES) 2003–2004 and comparisons with NHANES 1999–2000. *Environmental Health Perspectives* 115:1596–1602. DOI:10.1289/ehp.10598
- Chen H., M. Reinhard, V.T. Nguyen, and K.Y. Gin. 2015. Reversible and irreversible sorption of perfluorinated compounds (PFCs) by sediments of an urban reservoir. *Chemosphere* 144:1747–1753. DOI:10.1016/j.chemosphere.2015.10.055
- de Voogt P., U. Berger, W. de Coen, W. de Wolf, E. Heimstad, M. McLachlan, S. van Leeuwen, and A. van Roon. 2006. Perfluorinated organic compounds in the European environment. Perforce project NEST–508967. Scientific Report to the EU, University of Amsterdam, Amsterdam, Netherlands. 126 p.
- Furdui V.I., N.L. Stock, D.A. Ellis, C.M. Butt, D.M. Whittle, P.W. Crozier, E.J. Reiner, D.C.G. Muir, and S.A. Mabury. 2007. Spatial distribution of perfluoroalkyl contaminants in lake trout from the Great Lakes. *Environmental Science and Technology* 41:1554–1559. DOI:10.1021/es0620484
- Giesy J.P. and K. Kannan. 2001. Global distribution of perfluorooctane sulfonate in wildlife. *Environmental Science and Technology* 35:1339–42. DOI:10.1021/es001834k
- Gulkowska A., Q. Jiang, M.K. So, S. Taniyasu, P.K.S. Lam, and N. Yamashita. 2006. Persistent perfluorinated acids in seafood collected from two cities of China. *Environmental Science and Technology* 40:3736–3741. DOI:10.1021/es060286t
- Guo R., W.J. Sim, E.S. Lee, J.H. Lee, and J.E. Oh. 2010. Evaluation of the fate of perfluoroalkyl compounds in wastewater treatment plants. *Water Research* 44:3476–3486. DOI:10.1016/j.watres.2010.03.028
- Higgins C.P. and R.G. Luthy. 2006. Sorption of perfluorinated surfactants on sediments. *Environmental Science and Technology* 40:7251–7256. DOI:10.1021/es061000n
- Hoffman K., T.F. Webster, S.M. Bartell, M.G. Weisskopf, and T. Fletcher. 2011. Private drinking water wells as a source of exposure to fluorooctanoic acid (PFOA) in communities surrounding a fluoropolymer production facility. *Environmental Health Perspectives* 119:92–97. DOI:10.1289/ehp.1002503
- Kallenborn R., U. Berger, and U. Järnberg. 2004. Perfluorinated alkylated substances (PFAs) in the nordic environment. A TemaNord report of the Norwegian Institute for Air Research (NILU) (Kjeller, Norway) and the Institute for Applied Environmental Research (ITM), TemaNord 2004:552. Stockholm University, Stockholm, Sweden. 113 p.
- Loos R., J. Wollgast, T. Huber, and G. Hanke. 2007. Polar herbicides, pharmaceutical products, perfluorooctanesulfonate (PFOS), perfluorooctanoate (PFOA), and nonylphenol and its carboxylates and ethoxylates in surface and tap waters around Lake Maggiore in Northern Italy. *Analytic and Bioanalytical Chemistry* 387:1469–1478. DOI:10.1007/s00216–006–1036–7
- Martin J.W., D.M. Whittle, D.C.G. Muir, and S.A. Mabury. 2004. Perfluoroalkyl contaminants in a food web from Lake Ontario. *Environmental Science and Technology* 38:5379–5385. DOI:10.1021/es049331s
- Murakami M., H. Shinohara, and H. Takada. 2009a. Evaluation of wastewater and street runoff as sources of perfluorinated surfactants. *Chemosphere* 74:487–493. DOI:10.1016/j.chemosphere.2008.10.018
- Murakami M., K. Kuroda, N. Sato, T. Fukushima, S. Takizawa, and H. Takada. 2009b. Groundwater pollution by perfluorinated surfactants in Tokyo. *Environmental Science and Technology* 43:3480–3486. DOI:10.1021/es803556w
- Nakata H., K. Kannan, T. Nasu, H. Cho, E. Sinclair, and A. Takemura. 2006. Perfluorinated contaminants in sediments and aquatic organisms collected from shallow water and tidal flat areas of the Ariake Sea, Japan: Environmental fate of perfluorooctane sulfonate in aquatic ecosystems. *Environmental Science and Technology* 40:4916–4921. DOI:10.1021/es0603195
- Nguyen V.T., N. Reinhart, and J.Y. Karina. 2011. Occurrence and source characterization of perfluorochemicals in an urban watershed. *Chemosphere* 82:1277–1285. DOI:10.1016/j.chemosphere.2010.12.030
- Post G.L., J.B. Louis, K.R. Cooper, B.J. Boros–Russo, and R.L. Lippincott. 2009. Occurrence and potential significance if perfluorooctanoic acid (PFOA) detected in New Jersey public drinking water systems. *Environmental Science and Technology* 43:4547–4554. DOI:10.1021/es900301s
- Prudic D.E. 1991. Estimates of hydraulic conductivity from aquifer–test analyses and specific–capacity data, Gulf Coast regional aquifer systems, south–central United States. *Water Resources Investigation Report* 90–4121. U.S. Geological Survey, Austin, Texas, USA. 38 p.

- Saito N., K. Harada, K. Inoue, K. Sasaki, T. Yoshinaga, and A. Koizumi. 2004. Perfluorooctanoate and perfluorooctane sulfonate concentrations in surface water in Japan. *Journal of Occupational Health* 46:49–59. DOI:10.1539/joh.46.49
- Shultz M.M., D.F. Barofsky, and J.A. Field. 2004. Quantitative determination of fluorotelomer sulfonates in groundwater by LC MS/MS. *Environmental Science and Technology* 38:1828–1835. DOI:10.1021/es035031j
- Sinclair E. and K. Kannan. 2006. Mass loading and fate of perfluoroalkyl surfactants in wastewater treatment plants. *Environmental Science and Technology* 40:1408–1414. DOI:10.1021/es051798v
- Skutlarek D., M. Exner, and H. Farber. 2006. Perfluorinated surfactants in surface and drinking waters. *Environmental Science and Pollution Research* 13:299–307. DOI:10.1065/espr2006.07.326
- So M.K., Y. Miyake, W.Y. Yeung, Y.M. Ho, S. Taniyasu, P. Rostkowski, N. Yamashita, B.S. Zhou, X.J. Shi, J.X. Wang, J.P. Giesy, H. Yu, and P.K.S. Lam. 2007. Perfluorinated compounds in the Pearl River and Yangtze River of China. *Chemosphere* 68:2085–2095. DOI:10.1016/j.chemosphere.2007.02.008
- Tanaka S., S. Fujii, N.P.H. Lien, M. Nozoe, H. Fukagawa, W. Wirojanagud, A. Anton, and G. Lindstrom. 2006. A simple pre-treatment procedure in PFOS and PFOA water analysis and its' application in several countries. *Organohalogen Compounds* 68:527–530.
- van Leeuwen S.P.J., I. van der Veen, P.E.G. Leonards, and J. de Boer. 2006. Perfluorinated compounds in edible Dutch fish. *Organohalogen Compounds* 68: 535–538.
- Weremiuk A.M., S. Gerstmann, and F. Hartmut. 2006. Quantitative determination of perfluorinated surfactants in water by LC–ESI–MS/MS. *Journal of Separation Science* 29: 2251–2255. DOI:10.1002/jssc.200600041
- White S.S., S.E. Fenton, and E.P. Hines. 2011. Endocrine disrupting properties of perfluorooctanoic acid. *Journal of Steroid Biochemistry and Molecular Biology* 127:16–26. DOI:10.1016/j.jsbmb.2011.03.011
- Xiao F., M.K. Simcik, and J.S. Gulliver. 2011. Perfluoroalkyl acids in urban stormwater runoff: Influence of land use. *Water Research* 46:6601–6608. DOI:10.1016/j.watres.2011.11.029
- Yamashita N., K. Kannan, S. Taniyasu, Y. Horii, G. Petrick, and T. Gamo. 2005. A global survey of perfluorinated acids in oceans. *Marine Pollution Bulletin* 51:658–68. DOI:10.1016/j.marpolbul.2005.04.026

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Length–weight and length–length relationships of the Hogfish, *Lachnolaimus maximus*, off the northern coast of the Yucatan Peninsula, Mexico

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SHORT COMMUNICATION

LENGTH-WEIGHT AND LENGTH-LENGTH RELATIONSHIPS OF THE HOGFISH, *LACHNOLAIMUS MAXIMUS*, OFF THE NORTHERN COAST OF THE YUCATAN PENINSULA, MEXICO

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KEY WORDS: Southern Gulf of Mexico, Labridae, Campeche bank, Spear fishing

INTRODUCTION

The Hogfish, *Lachnolaimus maximus* (Walbaum, 1792), (Teleostei: Labridae) is a monandric, protogynous hermaphrodite wrasse (McBride and Johnson 2007) inhabiting structured reef habitats of the western North Atlantic Ocean, Gulf of Mexico (GOM), and the Caribbean Sea (Westneat 2002), where it has high economic importance (McBride and Murphy 2003) as a fishery resource. Off the Florida coast, it reaches a maximum size of 84 cm fork length (FL) and a maximum body weight of 10 kg (McBride and Richardson 2007). Due to heavy fishing pressure, there is evidence of an overfished condition for *L. maximus* (McBride and Murphy 2003, Ault et al. 2005) with population declines of 60%. Regionally, due to this condition it is classified as vulnerable by the International Union for Conservation of Nature (Choat et al. 2010); thus, it is important to record as much biological information as possible to understand its population dynamics in the region.

In Mexico, relatively nothing is known regarding the biology and fishery of *L. maximus* despite being intensively harvested by spearfishing in the Yucatan Peninsula (Sánchez—Aké and Medina—Quej 2008), yet there is an urgent need to implement fishery management strategies for this species. Its length—weight relationships (LWRs) have not been previously estimated from the southern GOM, and no record of any LWR from Mexico is available in Fishbase (Froese and Pauly 2014). Length—length relationships (LLRs) and LWRs are useful for conversion purposes when comparisons are made with literature values and consequently to estimate fish biomass (Ault et al. 2005, Aburto—Oropeza et al. 2011). The aim of this study was to describe the LWRs and LLRs of the *L. maximus* taken off the northern coast of the Yucatan Peninsula, Mexico, in the southern GOM.

MATERIALS AND METHODS

Recreational fishermen captured fish under clear water conditions with spear guns in shallow (5–10 m deep) coastal areas <5 km from shore between April and September each year from 2011–2013. Collection sites were off the coast of Chuburná (21°15'68N; 89°54'11W) and Chelem

(21°15'93N; 89°47'89W), on the northern coast of the Yucatan Peninsula, Mexico. Fish were measured to fork (FL), standard (SL), total length (TL, cm), and whole wet weight (W, g) was recorded. All LLRs (TL–FL, FL–SL, FL–TL, and SL–TL) were estimated using linear regressions on \log_{10} data. The LWRs were calculated as $\log_{10} W = \log_{10} a + b \log_{10} TL$, where “a” is the intercept of the regression curve (coefficient related to body form) and “b” is the regression coefficient (exponent indicating isometric growth) (Le Cren 1951, Froese 2006). Based on the slope (b) of the LWR, one can estimate whether fish growth is isometric ($b = 3$, all fish dimensions increase at the same rate), hypoallometric ($b < 3$) or hyperallometric ($b > 3$) (Froese 2006, Froese et al. 2011). Exploring which growth (i.e., isometric or allometric) is exhibited by a given species provides inference on how fish body proportions may vary at a given geographic area or during a specific season.

The b—value of each LWR, for each year, was verified by Student's t—test in order to determine if growth matched the isometric type ($b=3$), following the equation $t = (b-3)/S_b$, where t = Student t—value, b = slope, S_b = slope of standard error. Fish body size was compared between years with a one-way ANOVA ($\alpha=0.05$) and a Tukey HSD post-hoc test was used to identify differences. Regression slopes of the LWRs were compared between years (year as covariate) using an ANCOVA. Normality was determined with the Kolmogorov-Smirnov test, and homogeneity of variance was verified with the Bartlett's test. Regression analyses were carried out using Statgraphics Centurion, and performed according to Sokal and Rohlf (2012). Some obvious outliers were detected with the Dixon's test and removed from the regression analysis as recommended by Froese et al. (2011) using a criteria of line best fitting.

RESULTS AND DISCUSSION

A total sample of 292 fish was collected with specimens ranging from 16.6–36.5 cm FL (26.4 ± 0.23 se) and 94–842 g (392.3 ± 8.0 se) (Table 1). Significant differences were detected in fish size between years (ANOVA: $F_{2, 291} = 11.28$, $p <$

TABLE 1. Principal parameters and linear relations ($y = a + bx$) between fork length and whole wet weight per year, and combined (data pooled), for the Hogfish, *Lachnolaimus maximus*, from the northern coast of the Yucatan Peninsula, Mexico.

Year	Month								FL (cm), mean \pm se (min – max)	W (g), mean \pm se (min – max)	Regression Parameters		
	n	March	April	May	June	July	Aug	Sep			a (se)	b (se)	r ²
2011	56	0	0	7	0	15	31	3	25.6 \pm 0.54 (19.5–34.5)	395.6 \pm 18.5 (200–660)	–0.77 (0.11)	2.39 (0.08)	0.95
2012	107	38	12	25	17	0	15	0	27.8 \pm 0.36 (21.0–35.0)	448.3 \pm 14.2 (210–42)	–0.74 (0.08)	2.33 (0.05)	0.95
2013	129	0	0	0	12	37	40	40	25.6 \pm 0.33 (16.6–35.5)	344.6 \pm 11.3 (94–770)	–1.29 (0.05)	2.70 (0.04)	0.97
Total	292	38	12	32	29	52	86	43	26.4 \pm 0.23 (16.6–35.5)	392.3 \pm 8.2 (94–842)	–1.15 (0.05)	2.58 (0.03)	0.95

FL = fork length; W = fish wet weight; a = intercept value; b = regression slope; r² = coefficient of determination; se = standard error; n = number of fish sampled; min = minimum; max = maximum.

0.05), with larger fish found in 2012. Calculated LWRs were significantly different from zero ($p < 0.05$), with r^2 ranging from 0.95–0.97 (Table 1), which indicates strong model fits over the fish size range examined. Additionally, the LLRs calculated for *L. maximus* in this study had r^2 values ranging from 0.84–0.92 (Table 2). These equations can be useful for conversion purposes to calculate from a given length into another for data comparisons.

For *L. maximus* off the northern coast of the Yucatan Peninsula the exponent b in the LWRs (data pooled = 2.58; range = 2.33–2.70) was significantly lower than the isometric value ($b = 3$) each year (2011: $n = 56$; $t_s = -7.896$; 2012: $n = 107$, $t_s = -6.428$; 2013: $n = 129$, $t_s = -6.783$); thus, *L. maximus* exhibited hypoallometric growth. This means that the fish becomes less rotund as the growth increases, and also implies that either large specimens have changed their body shape to become more elongated or small specimens were in better nutritional condition at the time of sampling (Froese 2006). In fact, the LWR calculated for *L. maximus* off the northern Yucatan Peninsula was based on whole wet weight. Consequently, the slope in the regression line is less steep and low compared to that if the LWR was based on gutted weight. No previous growth type was ever calculated for *L. maximus* from the southern GOM; thus, this represents a first report.

There were significant differences between the yearly slopes of the LWRs (ANCOVA, $F_{5,291} = 245.9$, $p < 0.05$), which may be a reflection of growth variation, as LWR can fluctuate due to the influence of temperature, salinity, food availability, and reproduction (Froese et al. 2011). This may imply fish growth is experiencing variation according to food availability between years. However, it is possible such variations may be also be due to the influence of reproductive processes on individual growth (i.e., differential growth by sex), since *L. maximus* is hermaphroditic (McBride and Johnson 2007). Spawning is concentrated during winter and spring (December to May), peaking in April (McBride

and Johnson 2007) off the Florida coast. The reproductive season is still unknown for *L. maximus* in the southern GOM, and reproductive aspects were not explored in this study.

This work is the first study describing the LWRs and LLRs for *L. maximus* off the northern coast of the Yucatan Peninsula, southern GOM. The size range of *L. maximus* we examined (16.6–35.5 cm FL) was relatively lower than fish studied by Sánchez–Aké and Medina–Quej (2008; 26–54 cm FL) in Holbox, on the northeastern coast of the Yucatan Peninsula. Such differences may reflect fishery selectivity, variation in fishing seasons, and the influence of reproductive season. In Holbox, fish were speared from August to November but in our study they were speared from April to September. Distance to shore and depth in both locations (i.e., Holbox and our study area) were relatively similar. Additionally, the exponent b , calculated for *L. maximus* in the northern coast of the Yucatan Peninsula ($b = 2.50$), was similar to that calculated for specimens taken off Holbox ($b = 2.52$; Sánchez–Aké and Medina–Quej 2008), in the northeastern coast of the Yucatan Peninsula. This implies

TABLE 2. Linear relations ($y = a + bx$) between lengths (FL, TL, SL; data pooled) for the Hogfish, *Lachnolaimus maximus*, from the northern coast of the Yucatan Peninsula, Mexico.

Relationship	Regression Parameters			
	n	a (\pm se)	b (\pm se)	r ²
TL–FL	292	0.02 (0.02)	1.01 (0.17)	0.92
FL–SL	292	0.44 (0.03)	0.73 (0.02)	0.84
FL–TL	292	0.08 (0.02)	0.91 (0.01)	0.92
SL–TL	292	–0.25 (0.03)	1.08 (0.02)	0.90

FL = fork length; TL = total length; SL = standard length; a = intercept value; b = regression slope; r² = coefficient of determination; se = standard error; n = number of fish sampled.

that the environmental conditions (i.e., temperature, salinity) and food availability may be similar in both locations. Such differences may reflect growth rate variation as consequence of the prevailing environmental conditions in which *L. maximus* could be exposed to at geographic level. However, any differences in *b* would also depend on the weight used to estimate the LWR; in this particular case, whole wet weight of *L. maximus* was used in both locations.

Since *L. maximus* is recognized by the IUCN as vulnerable (Choat et al. 2010), it is imperative to record information not

only on its biology but also on fishery landings in order to estimate possible effects of overfishing. Unfortunately, the fishery status for *L. maximus* off the northern coast of the Yucatan Peninsula is unknown. However, data presented herein on length–weight relations of *L. maximus* in this region may be the first step in understanding regional biological aspects. Our study will be useful in providing some baseline data for fish stock assessments and population dynamics on this species in the region.

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LITERATURE CITED

- Aburto-Oropeza, O., B. Erisman, G.R. Galland, I. Mascareñas-Osorio, E. Sala, and E. Ezcurra. 2011. Large recovery of fish biomass in a no-take marine reserve. *PLoS One*: 6(8): e23601. DOI: 10.1371/journal.pone.0023601.
- Ault, J. S., S.G. Smith, and J.A Bohnsack. 2005. Evaluation of average length as an estimator of exploitation status for the Florida coral–reef fish community. *ICES Journal of Marine Science* 62:417–423. DOI: 10.1016/j.icesjms.2004.12.001.
- Choat, J.H., D. Pollard, and Y.J. Sadovy. 2010. *Lachnolaimus maximus*. The IUCN Red List of Threatened Species. Version 2015.2. www.iucnredlist.org. Downloaded on 26 June 2015.
- Froese, R. 2006. Cube law, condition factor and weight–length relationships: history, meta–analysis and recommendations. *Journal of Applied Ichthyology* 22:241–253. DOI: 10.1111/j.1439–0426.2006.00805.x.
- Froese, R. and D. Pauly (Eds.). 2014. FishBase. www.fishbase.org. Viewed on 10/06/2015
- Froese, R., A.C. Tsikliras, and K.I. Stergiou. 2011. Editorial note on weight–length relations of fishes. *Acta Ichthyologica et Piscatoria* 41:261–263. DOI: <http://dx.doi.org/10.3750/AIP2011.41.4.01>.
- Le Cren, E.D. 1951. The length–weight relationship and seasonal cycle in gonad weight and condition in the perch (*Perca fluviatilis*). *Journal of Animal Ecology* 20:201–219. DOI: 10.2307/1540.
- McBride, R.S. and M.D. Murphy. 2003. Current and potential yield per recruit of hogfish, *Lachnolaimus maximus*, in Florida. *Proceedings Gulf Caribbean Fisheries Institute* 54:513–525.
- McBride, R.S. and M.R. Johnson. 2007. Sexual development and reproductive seasonality of hogfish (Labridae: *Lachnolaimus maximus*), an hermaphroditic reef fish. *Journal of Fish Biology* 71:1270–1292. DOI: 10.1111/j.1095–8649.2007.01580.x
- McBride, R.S. and A.K. Richardson. 2007. Evidence of size selective fishing mortality from an age and growth study of hogfish (Labridae: *Lachnolaimus maximus*), a hermaphroditic reef fish. *Bulletin of Marine Science* 80:401–417.
- Sánchez-Aké, E.O. and A. Medina-Quej. 2008. Análisis de la pesquería de Boquinete, *Lachnolaimus maximus*, en la Isla de Holbox, Quintana Roo, México. *Proceedings Gulf and Caribbean Fisheries Institute* 60:287–296.
- Sokal, R. R. and F.J. Rohlf. 2012. Biometry: the principles and practice of statistics in biological research. 4th edition. W. H. Freeman and Co., New York, NY, USA, 937 p.
- Westneat, M. W. 2002. Labridae. In: Carpenter, K.E., ed. The Living Marine Resources of the Western Central Atlantic. Vol. 3: Bony Fishes, Part 2 (Opistognathidae to Molidae), Sea Turtles and Marine Mammals. FAO Species Identification Guide for Fishery Purposes and American Society of Ichthyologists and Herpetologists Special Pub. No. 5, Food Agriculture Organization, Rome, Italy, p. 1701–1722.

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SHORT COMMUNICATION

HABITAT USE AND VERTICAL DISTRIBUTION OF THE GREAT BARRACUDA *SPHYRAENA BARRACUDA* (EDWARDS 1771) IN THE WESTERN NORTH ATLANTIC USING ELECTRONIC ARCHIVAL TAGS

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KEY WORDS: barracuda, PSAT, behavior, movement, tagging

INTRODUCTION

The Great Barracuda, *Sphyraena barracuda* (Edwards 1771), is a large predatory teleost that inhabits tropical and subtropical environments worldwide, with the exception of the eastern Pacific Ocean (de Sylva 1963). Great Barracuda exhibit an ontogenetic shift in habitat as they mature. Juveniles typically spend the first two years in sheltered mangrove or seagrass habitats. By the winter of the second year, and approaching a size of *ca.* 300 mm total length (TL), they will leave these protected areas and associate more with offshore reefs and structure (de Sylva 1963, Blaber 1982). As adults, they inhabit the same trophic level as other large reef-associated predators (de Sylva 1963).

Although Great Barracuda are not often targeted by commercial fisheries as a food fish due to the threat of ciguatera poisoning, they do provide economic value for surrounding communities. Recreational fishermen and charter boats contribute to local economies through revenue generated by tourism (Granek et al. 2008), and Great Barracuda are also prized as a sportfish by some anglers because of the species' fighting ability and acrobatics (Sosin 2000). Even when they are not intentionally targeted, the historical catch data for Great Barracuda show that they are frequently caught incidentally (NOAA Fisheries 2015). However, Great Barracuda are not regulated at the federal level, and at the state level in Florida, they fall in the "unregulated" fishes category, thereby allowing fishermen to keep "two fish or 100 pounds of fish, whichever is greater" (FWC 2015a). Anecdotal observations of long-term declines in Great Barracuda abundance and catch rates, combined with public pressure from stakeholders, have caused the Florida state Fish and Wildlife Conservation Commission (FWC) to impose additional regulations of a recreational and commercial daily bag limit of two fish per person and a daily vessel limit of six fish per vessel as of 1 November 2015 (FWC 2015b).

Previous studies have documented that Great Barracuda are capable of travelling up to 12 km per day (O'Toole et al. 2011) and across the Gulf of Mexico (*ca.* 1000 km) over a period of about nine months (Villareal et al. 2007). O'Toole et al. (2010) also documented a depth of 32.22 meters using

limited ultrasonic telemetry in the Bahamas. In April and August 2014, two Great Barracuda were tagged off Miami, Florida with pop-up satellite archival tags (PSATs) to further describe the habitat use and depth preferences of this species.

MATERIALS AND METHODS

Specimens were caught by traditional hook-and-line recreational angling methods in the waters off Dade County, Florida. During the tagging event, gear type, handling time, hook location, and GPS coordinates of release location were recorded, in addition to fork length (FL, cm). Great Barracuda were also measured on a modified ACCESS scale to determine the overall condition of the specimen. The ACCESS scale is an overall health assessment of the fish with a qualitative grade from 0–10 using physical characteristics such as overall activity and general state of the musculature (see Kerstetter et al. 2003); potential fish required a minimum of 7 on this scale in order to be tagged. Fish were also not tagged if there were other large predators, such as sharks, visible in the vicinity of the tagging vessel.

During the tagging, the individual fish was brought on board the boat using a large landing net, then tagged and immediately returned to the water. A PSAT tag was inserted near the anterior dorsal fin to lock the dart head behind the pterygiophore bones. By anchoring through the dorsal musculature, the tag tether was well supported and the tag float provided sufficient lift to keep the tag body from rubbing against the body at slow speeds (Block et al. 1998). A conventional streamer tag was also inserted into the dorsal musculature on the opposite side of the fish from the PSAT.

Two Microwave Telemetry (Columbia, MD, USA) X-Tag HR model tags were used. The PSAT itself is contained in a composite, slightly positively buoyant, low-drag housing towed by a short monofilament leader attached to a medical-grade nylon dart; the entire tag and attachment tether combination weighed *ca.* 53 g. About 15 cm of 120 pound-test monofilament was used to attach the tag to a nylon dart anchor with aluminum crimps, and each crimped con-

nection was covered with electrical shrink tubing to help prevent chafing and minimize reflection. The monofilament segment also included a 150 pound—test ball bearing swivel (model BX4RZ, Sampo, Inc.; Barneveld, NY) to allow the tag to rotate freely and not incur rotational torque stress around the attachment site (Bain 2004, Kerstetter et al. 2011).

The PSATs were programmed to record a data point every 141 seconds. Water temperature ($^{\circ}\text{C}$ in 0.18° resolution), pressure (converted to depth in 1.3 m resolution), and light level (unit—less scale from 0–255) were recorded for 15 days. However, the X—tag HR model does not provide enough light level data for light—based geolocation estimates. All horizontal displacements were considered conservative based on the minimum straight—line distance (MSLD) between the location of tagging and the location of the first Argos transmission with < 1.5 km error (location class 1,

2, or 3). Survival was assessed through analysis of changes in the light, temperature, and depth data (see Kerstetter et al. 2011).

Data analysis

In order to evaluate diel changes in behavior, local sunrise and sunset time data were obtained from United States Naval Observatory (USNO; <http://www.usno.navy.mil>). Crepuscular periods were defined as 15 minutes before and after sunrise/sunset and were subsequently excluded from the diel analyses. USNO data were also used to assess day and night length to standardize habitat use. Variances were tested using an F—test, and found to be unequal. A two—sample t—test assuming unequal variances was used to evaluate diel differences over the 15—day deployment in the temperature and depth data, with significance assessed at $\alpha < 0.05$. Positions of the Florida Current and the Gulf Stream, as well as local sea surface temperatures (SSTs) during the deployment periods were assessed through satellite—derived imagery obtained from the Rutgers University Center for Ocean Observing Leadership (RUCOOL; <http://rucool.marine.rutgers.edu>).

RESULTS

Two Great Barracuda were caught using standard trolling techniques by a charter vessel operating out of Miami Beach, FL. The fish with PSAT 88094 measured 104 cm FL and was captured on 23 April 2014, while the fish with PSAT 88095 measured 101 cm FL and was captured on 17 August 2014. The hook was removed from the first individual, although it was determined that removal of the hook from the lower jaw in the other would have taken too long and harmed the fish further. Both scored a 9 on the ACCESS scale.

During the deployment periods, both fish travelled north. Tag 88094 was deployed at 25.702°N , 80.083°W , and travelled to 29.895°N , 80.548°W , *ca.* 100 km off the coast of northeast Florida (Figure 1). After detaching, tag 88094 transmitted 91% of its data. The MSLD for this fish was 471 km during the 15—day period, or a mean of 31.4 km per day. Recorded temperatures ranged from 18.6 – 28.4°C (Figure 2A). The maximum depth encountered by this tag was 145.2 m, with 90% of the deployment time being spent in water between 0 and 26.9 m (Figure 2B). Tag 88095 was deployed at 25.700°N , 80.083°W , and travelled to 35.297°N , 73.608°W , *ca.* 175 km off the coast of North Carolina (Figure 1). After detaching, tag 88095 transmitted 87% of its data. This individual travelled 1231 km MSLD during the 15—day deployment period, for a mean of 82 km per day. The temperature ranged from 17.8 – 31.7°C (Figure 2C). This Great Barracuda dove to a maximum depth of 186.9 m, but spent 90% of the time in water between 0 and 9.4 m deep (Figure 2D).



FIGURE 1. Tagging and first transmission locations for two pop-up satellite archival tags (PSATs) deployed on Great Barracuda *Sphyrna barracuda* in April 2014 (88094) and August 2014 (88095) in the western North Atlantic Ocean. Tag 88094 travelled 471 km to the north-northwest and tag 88095 travelled 1231 km to the north-northeast during the 15-day PSAT deployment periods (lines denote approximate minimum straight-line distances, not actual tracks). The dashed line represents the Florida Current south of Cape Canaveral, Florida.

Both tagged Great Barracuda travelled north for the entire 15-day deployment duration, presumably following the mainstem of the Gulf Stream current. If either fish had died and the combined tag–carcass just drifted north in the current, the depth and temperature would be expected to stay somewhat unchanged. However, both individuals are thought to have stayed alive for the entire deployment duration because the depth and temperature values were constantly changing within diel patterning until the tags detached.

Both tag datasets were also examined for potential differences in behavior related to diel patterns. Great Barracuda 88094 showed a significant diel difference in overall time at depth ($t_{4901} = 25.35$, $p < 0.0001$). Daytime depth averaged 7.03 m and ranged from 0–86.1 m. Nighttime depth averaged 15.6 m and ranged from 0–145.2 m (Figure 2B). There was also a significant difference in time at temperature ($t_{7525} = 5.85$, $p < 0.0001$). Mean daytime temperature was 26.1°C and ranged from 20.4–28.4°C, while the mean nighttime temperature was 25.9°C and ranged from 18.6–27.7°C (Figure 2A). Great Barracuda 88095 also showed a significant diel difference in overall time at depth and temperature ($t_{5533} = 9.37$, $p < 0.0001$; $t_{6461} = 11.29$, $p < 0.0001$, respectively).

Mean daytime depth was 4.3 m and ranged from 0–72.6 m. Mean nighttime depth was 6.4, and ranged 0–186.9 m (Figure 2D). Mean daytime temperature was 30.1°C, and ranged from 24.9–31.7°C, while mean nighttime temperature was 29.8°C and ranged from 17.8–31.7°C (Figure 2C).

Detailed analyses of daily temperature use patterns for both tagged fish indicated a change in local water conditions; temperature–at–depth profiles indicated this transition on 3 May for 88094 and 27 August for 88095. Daily comparisons of ambient surface temperature data from the PSAT records and the satellite–derived SST data from RUCOOL strongly suggest movements by both tagged fish from the warmer waters of the Gulf Stream to somewhat cooler inshore waters at the end of both deployments.

DISCUSSION

The two tagged individuals from this study are the only documented use of PSAT technology on the Great Barracuda, and thus represent the only long–duration behavior data available for this species. However, several previous studies have examined the short–duration movements of Great Barracuda. For example, Springer and McErlean (1961) used conventional tags in the Florida Keys to demon-

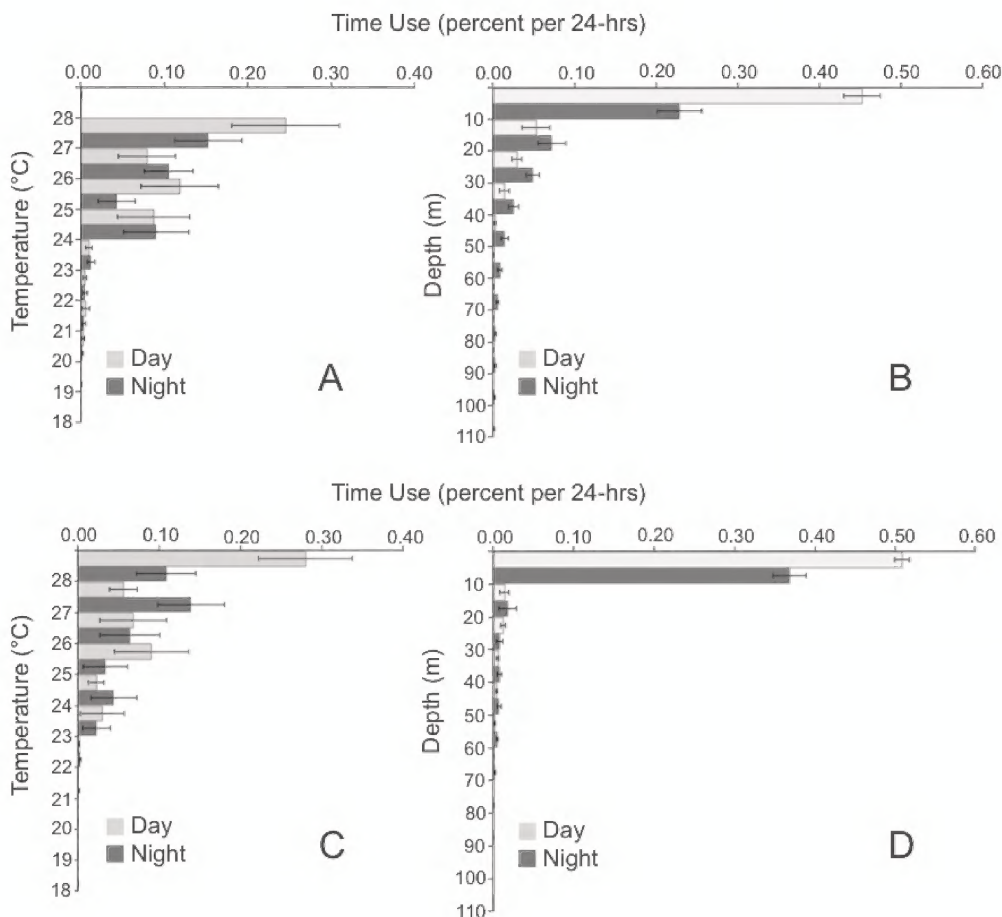


FIGURE 2. Time-at-temperature and time-at-depth histograms from two Microwave Telemetry HR model X-Tags deployed on Great Barracuda *Sphyræna barracuda* in South Florida during April and August 2014. A. Temperature, Fish 88094. B. Depth, Fish 88094. C. Temperature, Fish 88095. D. Depth, Fish 88095. Both day (light grey) and night (dark grey) results are presented as mean time use per 24-hour period over each 15-day deployment period, and error bars are ± 1 SE. Although both individual fish occasionally undertook short-duration movements to depths > 100 m, they constituted a small fraction of the total time utilization (0.6% for 88094 and 0.05% for 88095) and those depths are thus not shown.

strate short-term displacements of up to 193 km over a period of 90 days. Villareal et al. (2007) also used conventional tags to show that Great Barracuda are capable of travelling over 1000 km across the Gulf of Mexico over a period of years. Similarly, O'Toole et al. (2011) used an acoustic receiver array and tagged Great Barracuda in the Bahamas to show that they are capable of travelling 12 km per day and over 100 km to other islands in the Bahamas. However, the archived data and longer deployment durations of PSATs allow for a better description of Great Barracuda movement and depth preferences in much finer detail.

Both tagged individuals spent the majority of the time in the upper portion of the water column during daylight hours. Previous studies have shown that Great Barracuda are visual feeders (de Sylva 1963; Porter and Motta 2004). Great Barracuda may be opportunistically feeding near the surface of the water during daylight hours, as the prey items found in mature individuals in the previous studies (e.g., Clupeidae, Exocoetidae) also inhabit the upper region of the water column (de Sylva 1963; Randall 1967). There may also be a thermal advantage to remaining in the warmer surface waters between foraging in colder depths, a behavior seen in other large pelagic fishes (e.g., White Marlin *Kajikia albida* in Horodysky et al. 2007). However, Great Barracuda have also been observed preying on some reef-associated bottom-dwelling species (e.g., Holocentridae, Balistidae, Scaridae; Hansen 2015). Although the times at temperature for both individuals tagged in this study were significantly higher during the day, it is hard to imagine that these differences are meaningful. More likely, these differences are an artifact of vertical movements rather than temperature preference. These infrequent vertical movements to depth during both daytime and nighttime hours could be associated with feeding on deeper-dwelling prey items, as have other fishes that are generally considered epipelagic predators (e.g., Sailfish *Istiophorus platypterus* in Kerstetter et al. 2011; Shortfin Mako *Isurus oxyrinchus* in Loefer et al. 2005). An increase in the number of PSAT tagged fish species is allowing a better understanding of habitat use and segregation, and the results from studies such as this one should be combined to investigate behavioral patterns on an ecosystem scale (Block et al. 2011).

The results from this study clearly complement prior works. Great Barracuda 88094 travelled over 471 km MSLD, while individual 88095 traveled 1231 km MSLD in the 15-day deployment period. While it has been shown that Great Barracuda are capable of moving great distances (Springer and McErlean 1961; Villareal et al. 2007; O'Toole et al. 2011), this much movement in such a short time span was unexpected. Although de Sylva (1963) suggested that Great Barracuda may migrate northward in the spring, such migration has not been documented by any other sources.

Regardless, this migratory pattern would not explain the movements from tag 88095, as it travelled north during the late summer months when de Sylva (1963) speculated that Great Barracuda would be moving southward. While none of the previous studies showed individual Great Barracuda traveling as far as in the present study, Villareal et al. (2007) did show that they were capable of displacement across the Gulf of Mexico within the span of a year. O'Toole et al. (2010) is one of the only studies to scientifically document the depth of Great Barracuda, although the maximum observed depth in that acoustic telemetry study was only 32.2 m, as compared with the 145.2 and 186.9 m depths observed with PSATs. The Great Barracuda tagged in the O'Toole et al. (2011) study ranged in size from 62 to 120 cm TL, and would thus be considered mature specimens. It is likely that the movements to the shelf habitats as noted by O'Toole et al. (2011) represent a different forage base for Great Barracuda, with accordingly different behaviors.

A larger question regards the presumption of local residency by Great Barracuda. Contrary to the observations in Wilson et al. (2006), neither of the tagged individual Great Barracuda in the present study remained within the vicinity of the tagging location. Kerstetter and Graves (2006), among others, have long noted the occurrence of large Great Barracuda in the offshore pelagic longline fishery, which may suggest a complex behavior combining local residency and long-range movement. The association of this species with the upper water column should allow the derivation of geolocation estimates via light-based algorithms in other PSAT models, especially when these estimates are refined with ambient SST data (e.g., Lam et al. 2010). Additional electronic and conventional tagging efforts, including molecular genetic analyses that expand upon the work by Daly-Engel et al. (2012), are recommended to better elucidate the coastal and pelagic movement patterns within Great Barracuda populations. If the species consists of multiple stocks or regularly moves between neighboring state waters, then management of Great Barracuda should occur at the federal level instead of by the individual states.

CONCLUSION

This study has shown that Great Barracuda are capable of traveling great distances (> 80 km per day) in a short time span. It also expands the habitat utilization understanding of Great Barracuda by documenting movements to depths greater than 140 m, and that these movements are often short and relatively deep. Combining these data with results from other oceanic predators could also show how Great Barracuda interact with more economically valuable targeted species, which could be beneficial when implementing future management strategies.

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LITERATURE CITED

- Bain, M.B. 2004. "Electronic Tags". Stock Identification Methods, Applications in Fishery Science. Elsevier Academic Press; San Diego, CA, USA, 736 p.
- Blaber, S.J.M. 1982. The ecology of *Sphyrna barracuda* (Osteichthyes: Perciformes) in the Kosi system with notes on the Sphyrnidae of other Natal estuaries. South African Journal of Zoology 17:171–176. DOI:10.1080/02541858.1982.11447799
- Block, B.A., H. Dewar, C. Farwell, and E.D. Prince. 1998. A new satellite technology for tracking the movements of Atlantic bluefin tuna. Proceedings of the National Academy of Sciences of the United States of America 95:9384–9389. DOI:10.1073/pnas.95.16.9384
- Block, B.A., I.D. Jonsen, S.J. Jorgensen, A.J. Winship, S.A. Shaffer, S.J. Bograd, E.L. Hazen, D.G. Foley, G.A. Breed, A.L. Harrison, and J.E. Ganong. 2011. Tracking apex marine predator movements in a dynamic ocean. Nature 475:86–90. DOI:10.1038/nature10082
- Daly–Engel, T.S., J.E. Randall, and B.W. Bowen. 2012. Is the Great Barracuda (*Sphyrna barracuda*) a reef fish or a pelagic fish? The phylogeographic perspective. Marine Biology 159:975–985. DOI:10.1007/s00227–012–1878–9
- De Sylva, D.P. 1963. Systematics and life history of the great barracuda, *Sphyrna barracuda* (Walbaum). Studies in Tropical Oceanography 1:1–179. http://scholarlyrepository.miami.edu/cgi/viewcontent.cgi?article=1014&context=trop_ocean
- FWC. 2015a. Unregulated Species, Florida Fish and Wildlife Conservation Commission. <http://myfwc.com/fishing/saltwater/recreational/unregulated-species/>. (viewed on 08/02/2015).
- FWC. 2015b. FWC approves barracuda conservation measures for South Florida. <http://myfwc.com/news/news-releases/2015/september/04/barracuda>. (viewed on 01/04/2016).
- Granek, E.F., E.M. Madin, M.A. Brown, W. Figueira, D.S. Cameron, Z. Hogan, G. Kristianson, P. de Villiers, J.E. Williams, J. Post, and S. Zahn. 2008. Engaging recreational fishers in management and conservation: Global case studies. Conservation Biology 22:1125–1134. DOI:10.1111/j.1523–1739.2008.00977.x
- Hansen, N.R. 2015. Feeding Ecology and Habitat Utilization of the Great Barracuda *Sphyrna barracuda* (Edwards 1771) in Southeast Florida, M.S. thesis. Nova Southeastern University, Dania Beach, FL USA, 56 p. http://nsuworks.nova.edu/occ_stuetd/32/
- Horodysky, A. Z., D.W. Kerstetter, R.J. Latour, and J.E. Graves. 2007. Habitat utilization and vertical movements of white marlin (*Tetrapturus albidus*) released from commercial and recreational fishing gears in the western North Atlantic Ocean: inferences from short duration pop-up archival satellite tags. Fisheries Oceanography 16:240–256. DOI:10.1111/j.1365–2419.2007.00419.x
- Kerstetter, D.W., B.E. Luckhurst, E.D. Prince, and J.E. Graves. 2003. Use of pop-up satellite archival tags to demonstrate survival of blue marlin (*Makaira nigricans*) released from pelagic longline gear. Fishery Bulletin 101:939–948. <http://fishbull.noaa.gov/1014/22kerste.pdf>
- Kerstetter, D.W. and J.E. Graves. 2006. Effects of circle versus J-style hooks on target and non-target species in a pelagic longline fishery. Fisheries Research 80:239–250. DOI:10.1016/j.fishres.2006.03.032
- Kerstetter, D.W., S.M. Bayse, J.L. Fenton, and J.E. Graves. 2011. Sailfish habitat utilization and vertical movements in the southern Gulf of Mexico and Florida Straits. Marine and Coastal Fisheries: Dynamics, Management, and Ecosystem Science 3:353–365. DOI:10.1080/19425120.2011.623990
- Lam, C.H., A. Nielsen, and J.R. Sibert. 2010. Incorporating sea–surface temperature to the light-based geolocation model TrackIt. Marine Ecology Progress Series 419:71–84. DOI:10.3354/meps08862
- Loefer, J.K., G.R. Sedberry, and J.C. McGovern. 2005. Vertical movements of a shortfin mako in the Western North Atlantic as determined by pop-up satellite tagging. Southeastern Naturalist 4:237–246. DOI:10.2307/3877960
- NOAA Fisheries. 2015. NOAA Office of Science and Technology, National Marine Fisheries Service. Recreational and Commercial Fisheries Statistics. <http://www.st.nmfs.noaa.gov/st1/index.html>. (viewed on 07/17/2013).
- O'Toole, A.C., K.J. Murchie, C. Pullen, K.C. Hanson, C.D. Suski, A.J. Danylchuk, and S.J. Cooke. 2010. Locomotory activity and depth distribution of adult great barracuda (*Sphyrna barracuda*) in Bahamian coastal habitats determined using acceleration and pressure biotelemetry transmitters. Marine and Freshwater Research 61:1446–1456. DOI:10.1071/MF10046
- O'Toole, A.C., A.J. Danylchuk, T.L. Goldberg, C.D. Suski, D.P. Philipp, E. Brooks, and S.J. Cooke. 2011. Spatial ecology and residency patterns of adult great barracuda (*Sphyrna bar-*

- racuda*) in coastal waters of The Bahamas. *Marine Biology* 158:2227–2237. DOI:10.1007/s00227-011-1728-1
- Porter, H.T. and P.J. Motta. 2004. A comparison of strike and prey capture kinematics of three species of piscivorous fishes: Florida gar (*Lepisosteus platyrhincus*), redfin needlefish (*Strongylura notata*), and great barracuda (*Sphyræna barracuda*). *Marine Biology* 145: 89–1000. DOI:10.1007/s00227-004-1380-0
- Randall, J.E. 1967. Food habits of reef fishes of the West Indies. *Studies in Tropical Oceanography* 5:665–847.
- Sosin, M. 2000. *The Complete Book of Light-tackle Fishing*. The Lyons Press, New York, NY USA, 224 p.
- Springer, V.G. and A.J. McErlean. 1961. Tagging of great barracuda *Sphyræna barracuda* (Walbaum). *Transactions of the American Fisheries Society* 90:497–500. DOI:10.1577/1548-8659(1961)90[497:TOGBSB]2.0.CO;2
- Villareal, T.A., S. Hanson, S. Qualia, E.L.E. Jester, H.R. Granade, and R.W. Dickey. 2007. Petroleum production platforms as sites for the expansion of ciguatera in the northwestern Gulf of Mexico. *Harmful Algae* 6:253–259. DOI:10.1016/j.hal.2006.08.008
- Wilson, S.K., D.T. Wilson, C. Lamont, and M. Evans. 2006. Identifying individual great barracuda *Sphyræna barracuda* using natural body marks. *Journal of Fish Biology* 69: 28–932. DOI:10.1111/j.1095-8649.2006.01128.x
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